5-1-2013

CLIMATE CHANGE, SHIFTS IN TROPICAL STORM REGIMES AND TRIADICA SEBIFERA INVASION IN COASTAL MISSISSIPPI, UNITED STATES

Shishir Paudel
Southern Illinois University Carbondale, paudelshishir@gmail.com

Recommended Citation
Paudel, Shishir, "CLIMATE CHANGE, SHIFTS IN TROPICAL STORM REGIMES AND TRIADICA SEBIFERA INVASION IN COASTAL MISSISSIPPI, UNITED STATES" (2013). *Dissertations.* Paper 694.

CLIMATE CHANGE, SHIFTS IN TROPICAL STORM REGIMES AND *TRIADICA SEBIFERA* INVASION IN COASTAL MISSISSIPPI, UNITED STATES

By

SHISHIR PAUDEL

M. S., University of Bergen, Norway, 2006
M. Sc., Tribhuvan University, Kirtipur, Kathmandu, Nepal, 2001

A Dissertation
Submitted in Partial Fulfillment of the Requirements for the
Doctor of Philosophy in Plant Biology

Department of Plant Biology
In the Graduate School
Southern Illinois University Carbondale
May, 2013

DISSERTATION APPROVAL

CLIMATE CHANGE, SHIFTS IN TROPICAL STORM REGIMES AND *TRIADICA SEBIFERA* INVASION IN COASTAL MISSISSIPPI, UNITED STATES

By
SHISHIR PAUDEL

A Dissertation Submitted in Partial
Fulfillment of the Requirements
for the Degree of
Doctor of Philosophy
In the field of Plant Biology
Approved by:

Dr. Loretta L. Battaglia, Chair
Dr. Dale H. Vitt
Dr. David J. Gibson
Dr. John W. Groninger
Dr. Sara G. Baer

Graduate School
Southern Illinois University Carbondale
March, 2013

AN ABSTRACT OF THE DISSERTATION OF

Shishir Paudel, for the Doctor of Philosophy degree in **PLANT BIOLOGY**, presented on **December 14th, 2012**, at Southern Illinois University Carbondale.

TITLE: CLIMATE CHANGE, SHIFTS IN TROPICAL STORM REGIMES AND *TRIADICA SEBIFERA* INVASION IN COASTAL MISSISSIPPI, UNITED STATES

MAJOR PROFESSOR: Dr. Loretta L. Battaglia

Global climate change is predicted to affect timing and severity of disturbance events (e.g., fire, drought, hurricanes, wind storms, and inundation), but the extent of these disturbance events and their impacts on natural ecosystems may vary regionally. Rising sea level, increased frequency and intensity of tropical storms, and altered inundation regimes are likely to create changing environmental conditions in low-lying coastal ecosystems. These large scale disturbances may increase resource availability and regeneration spaces, reduce competition, and possibly increase community vulnerability to invasion. Shifting disturbance regimes and invasion together are predicted to drive long-term shifts in coastal plant community structure and ecosystem processes. However, impacts of altered environmental conditions on native and invasive plant species and the species responses to changed environmental conditions are poorly understood. The aims of this study were: (i) to assess the probability of occurrence of juveniles of the invasive exotic *Triadica sebifera* and co-occurring native species, *Baccharis halimifolia, Ilex vomitoria*, and *Morella cerifera* in the field in relation to surrounding environmental factors, (ii) to assess the effects of elevated salinity across a typical coastal transition on germination of

T. sebifera, *B. halimifolia* and *M. cerifera*, using controlled growth chamber and greenhouse experiments, (iii) to assess the effects of climate change and shifting inundation and tropical storms regimes on *T. sebifera*, *B. halimifolia* and *M. cerifera*, and (iv) to evaluate the role of vesicular arbuscular mycorrhizae (VAM) on spread of invasive *T. sebifera* in coastal transition ecosystems at the Grand Bay National Estuarine Research Reserve (GBNERR), Coastal Mississippi, southeastern USA. Results from assessing the probability of occurrence of juveniles of invasive and co-occurring native species showed that soil water conductivity (i.e., salinity) was the major factor related to the occurrence of invasive *T. sebifera* and native *B. halimifolia, I. vomitoria* and *M. cerifera* along the coastal transitions. Probability of occurrence of the invasive *T. sebifera* was significantly related to landscape factors and occurrence was highest in close proximity to roads, trails, power lines, and recreational sites, and water bodies. These results imply that future increases in salinity will negatively impact *I. vomitoria*, *M. cerifera*, and *T. sebifera*, leading to range contraction of these species away from the coast. However, natural and anthropogenic disturbances that often increase resource pulses and reduce competition, likely increase the dominance of *T. sebifera* in already invaded areas. Positive effects of landscape structures on *T. sebifera* occupancy highlight the role of landscape variables in promoting new invasions in coastal forests of the southeastern USA. Controlled growth chamber and greenhouse germination experiments demonstrated that germination of all species (i.e., *T. sebifera*, *B. halimifolia*, and *M. cerifera*) decreased with elevated salinity and that the reduction was most pronounced in soils from the most seaward zones along the coastal transition. Although native *B. halimifolia* was least sensitive to elevated salinity, invasive *T. sebifera* displayed plasticity of germination trait across different salinity levels in most inland soils. These results suggest that the phenotypic plasticity may facilitate spread of *Triadica sebifera* under some degree of salinity stress in more inland section of the coastal transition. A manipulative greenhouse experiment demonstrated that simulated canopy damage from intense hurricane winds and associated storm surge produced differential effects on survival and growth of native (*B. halimifolia* and *M. cerifera*) and invasive (*T. sebifera*) species at simulated different forest conditions common in the GBNERR. Invasive *T. sebifera* was by far the most shade tolerant of the three species and seedling survival under highly shaded conditions may provide it with a competitive edge over native species during community reassembly following tropical storms. *T. sebifera* may better utilize post-hurricane conditions (e.g., resource-rich empty spaces) and potentially increase its dominance in coastal forested ecosystems. The last experimental study revealed that invasive *T. sebifera* had higher VAM colonization compared to co-occurring native species both in controlled greenhouse and field experiments, and that the higher colonization leads to significant increases in aboveground biomass, supporting the hypothesis that VAM fungi strongly benefit the invasive species. These results suggest that the VAM colonization may be necessary for the initial establishment of *T. sebifera* along the coastal transitions. Furthermore, my research also suggested that *T. sebifera* was not allelopathic and did not interfere with growth of native species. Overall, the findings of this research provide insight into the impacts of climate change related shifts on performance of invasive and co-occurring native species across coastal transitions of the southeastern USA. Variation in invasive and co-occurring native species' performances under changed environmental conditions (e.g., elevated salinity and increased light availability) and improved mutualistic association between invasive *T. sebifera* and VAM fungi may drive increased invasion with frequent community reassembly of low-lying coastal ecosystems undergoing rapid climate change.

ACKNOWLEDGEMENTS

I am greatly indebted to my advisor Dr. Loretta Battaglia for her tremendous support, encouragement, and tireless guidance in the development of this research project, field work, data analysis, the writing process, and helping me make it to the finish line. Her experience and guidance have improved my understanding of many aspects in the field of ecology. I would like to thank my other committee members Dr. Sara G. Baer, Dr. David J. Gibson, Dr. Dale H. Vitt, and Dr. John Groninger for their continuous support. I am thankful to SIU- Carbondale and the Department of Plant Biology for support through graduate assistantships. I would like to extend my sincere thanks to Dr. Mark Woodrey and the Grand Bay National Estuarine Research Reserve (GBNERR) for on-site support.

I am thankful to the many people who have provided invaluable help and support throughout this process and this work would not have been possible without their help. I am thankful to the Plant Biology Greenhouse staffs and John Miller at the Horticulture Research Center Greenhouse, SIU- Carbondale. I had a lot of help with my field and lab work from the Battaglia lab. Many thanks to Matt Abbott, Jesse Fruchter, Jason Inczauskis, Hannah Kalk, Brooklyn Krings, Jerrod Looft, Chelsey Hiller, Guy Schmale, Amanda Thalhammer, and Anthony Tate. Special thanks go to my fellow graduate students, Bin Xu and Juan Benavides, for providing friendship, good discussion, and essential comic relief.

Last, but not least, I am extremely grateful to my parents and family, my wife Urmila and son Bhawit, for their love, patience, understanding, and the support they have given me throughout my academic career. I only hope I can be as supportive in their future endeavors.

TABLE OF CONTENTS

LIST OF TABLES

CHAPTER 5

LIST OF FIGURES

CHAPTER 1

INTRODUCTION

Effects of Climate Change on Natural Ecosystems-

The Earth’s climate is changing because of increased greenhouse gases, rising temperatures, and changes in precipitation patterns (IPCC 2007a, Solomon et al. 2009). Global climate change is affecting individuals, populations, and ecosystems in many ways. Several plant and animal species with diverse geographical distributions have already been affected by climate change (Parmesan 2006, Walther 2010). Recent studies have suggested that climate change produces species range shifts that are poleward in latitude and upward in elevation (Parmesan 1996, 2006). A meta-analysis of 1,700 species revealed globally coherent fingerprints of climate change in range shifts (average 6.1 km per decade) towards higher latitudes or higher elevation (Parmesan and Yohe 2003). Another meta-analysis of 143 studies demonstrated changes in the distribution and range shifts of species, from mollusks to mammals and from grasses to trees (Root et al. 2003). Field and experimental studies showed increased abundance of shrub vegetation across northern Alaska, Canada, and parts of Russia (Sturm et al. 2001). Recent simulation models (using both general circulation models (GCMs)-HadAM3H and ECHAM4/OPYC3) suggested the shifts in regional climatic zones are likely to lead to changes in distribution and composition patterns of dominant tree species of Swedish forests (Koca et al. 2006). Phenological changes of plants and animal species in response to climate change have already been reported across many regions (Parmesan 2006, Walther 2010).

Climate change can affect occurrence, timing, duration, and intensity of extreme disturbance events including storm severity (Turner et al. 1998b, Dale et al. 2001). Disturbances are key processes that can affect species diversity (Huston 1979, Miller et al. 2011), community structure (Hobbs and Huenneke 1992, Lloret et al. 2012), and ecosystem functions (Foster et al. 1998). Natural disturbances, such as wildfires, drought, hurricanes, windstorms, landslides, coastal flooding, and insect and pathogen outbreaks can change the structure and functions of ecosystems over short time periods (Turner et al. 1998a, Dale et al. 2001, Scheffer et al. 2001). Extreme and altered environmental conditions may increase large-scale tree mortality and forest dieback (Michener et al. 1997, Dale et al. 2000, Anderegg et al. 2012). Large scale tree mortality, defoliation, and loss of canopy trees may increase the open spaces readily colonized by species that are better adapted to changing environmental conditions (Overpeck et al. 1989). Additionally, disturbances can act adversely in native plant communities by promoting invasions by exotic and weedy plant species (Rejmanek 1989, Hobbs and Huenneke 1992). In particular, the shifts in historical disturbance regimes and increase in frequency and intensity of disturbance, due to climate change, may promote establishment of invasive species by replacing native species (Hobbs and Huenneke 1992). Disturbance events may also increase nutrient and resource availability, which favor exotic invasive over native species (Davis et al. 2000, Blumenthal 2006). For instance, high-intensity fires that decrease canopy cover and increase light availability led to increases in alien plant species richness and dominance in the southern Sierra Nevadas (Keeley et al. 2003).

Effects of Climate Change on Coastal Ecosystems of the Southeastern United States-

Many low-lying coastal ecosystems around the world have evolved under the regimes of catastrophic disturbance by hurricanes, sea-level rise and fall, and their interactions (Michener et al. 1997, Greening et al. 2006). The coastal ecosystems of the southeastern United States (US) along the northern Gulf of Mexico are linked to a significant portion of the nation's economy and harbors one of the largest ecological systems in the country (Ning et al. 2003). These are low-elevation coastal systems experiencing subsidence and higher rate of relative sea level rise (2 to 10 mm per year; probably exacerbated by subsidence, reduced sedimentation, and soil and gas extraction) compared to global mean sea level rise (1.7 mm per year) (Titus et al. 2009). Although the magnitude of climate induced sea level rise remains highly uncertain (Nicholls and Cazenave 2010), recent climate change models predict a persistent rise in sea level with continued anthropogenic global warming and glacial ice sheets melting (IPCC 2007a). On the basis of the correlation between historical sea level rise and future global temperature scenarios of the IPCC, recent model projections for future sea level rise suggested a 0.75-1.90 m rise by the end of 21st century (Vermeer and Rahmstorf 2009). Additionally, as climate change continues, the intensification of tropical storms is likely to increase (Emanuel 2005; Webster et al. 2005) and the anthropogenic global warming may further fuel the increase in intense Atlantic hurricanes throughout this century (Bender et al. 2010). Additionally, rising sea levels may increase the hurricane flood risk at low elevation coastal areas through greater inland penetration by hurricane generated storm surges (Mousavi et al. 2011). Thus, coastal ecosystems of southeastern US along the northern Gulf of Mexico are at the forefront of climate change and are directly impacted by tropical storms and sea level rise (Michener et al. 1997, Scavia et al. 2002, Battaglia et al. 2012).

Hurricanes are one of the most powerful natural disturbances that influence plant community structure and composition (Michener et al. 1997, Turner et al. 2003) through sudden and massive tree mortality, successional change, and high species turnover (Lugo 2000). In low-lying coastal areas hurricane generated wind storms cause substantial damage to inland forested wetlands by uprooting trees, snapping boles, and defoliation (Foster 1988, Michener et al. 1997, Chambers et al. 2007), thereby altering physical environments through creation of canopy gaps and increase forest floor microsite heterogeneity (Battaglia et al. 1999). Scavia et al. (2002) predicted that a warmer sea surface increase hurricane wind strength and wind damage by up to 25%. Damage to the forest canopy results in greater light availability to understory vegetation, which promotes seed germination, regeneration, and accelerated seedling growth (Denslow 1987, Peterson and Pickett 1995). Increased rates of forest disturbance also promotes fast growing and disturbance tolerant plant species on the landscape (Overpeck et al. 1990). However, hurricane generated wind action can differentially affect coastal forested systems depending on location relative to wind direction, community structure, and degree of anthropogenic influence (Grove et al. 2000, Kupfer et al. 2008). Tree canopy damage often exposes inland plants to salt through salt spray and storm surges (Michener et al. 1997). Storm surges periodically submerge and expose the coastal communities to acute salinity stress which may persist in the soil for long time following intense hurricanes (Howard 2012). Long term inundation and salt stress alters community structure by causing defoliation and mortality of salt and flood-intolerant species (cf. Williams et al. 1999) and permitting the colonization of vacant sites by more stress-tolerant species which may bring shifts in successional directions in the community (Lugo 2000).

In addition to immediate impacts of recurrent tropical storms, sea level rise may have slow but enduring effects on coastal plant communities. Rising seas push highly saline water inland, causing a variety of impacts to low-lying coastal areas, including inundation, soil erosion, increased soil water salinity, and wrack deposition (Michener et al. 1997, IPCC 2007a, Titus et al. 2009, Tate and Battaglia 2012). These events eventually affect species distributions and community dynamics along the coastal transitions (Scavia et al. 2002, Mitsch and Gosselink 2007) by influencing germination and growth (Kozlowski 1997). As the rate of sea level rise accelerates, salinity sensitive and salt intolerant species may be lost or replaced by less sensitive and more tolerant novel species (Saha et al. 2011). For instance, low marsh cordgrass (*Spartina alterniflora*) is moving landward at the expense of high marsh species on the coast of New England, USA (Donnelly and Bertness 2001), and the salt tolerant invasive *Phragmites australis* is successfully colonizing and replacing native species in many coastal areas of the eastern USA (Vasquez *et al*. 2005).

Coastal wetlands in the coastal plain of Mississippi are already experiencing sea level rise and intense tropical storms (Battaglia et al. 2012). These systems harbor one of the most diverse ecosystems in the southeastern US (Harper 1913). In particular, a coastal estuarine reserve, The Grand Bay National Estuarine Research Reserve (GBNERR) lies in southeastern Mississippi near the border of Alabama and supports a biologically diverse estuarine ecosystems in the Northern Gulf of Mexico region (Hilbert 2006). The reserve encompasses coastal bay, salt marshes, brackish marshes, maritime pine forests, and pine savanna. Coastal vegetation of this region is vulnerable to climate change *vis a vis* intermittent hurricanes, storms, continuous sea-level rises, and flooding (Michener et al. 1997, Ning et al. 2003). The effects of hurricane disturbances on coastal ecosystems (e.g., changes in stand structure, species composition, decomposition etc.) have widely been recognized and studied extensively (Zimmerman et al. 1994, Battaglia et al. 1999, Chambers et al. 2007, Stanturf et al. 2007, Kupfer et al. 2008, Lugo 2008). Studies are beginning to emerge which address the social and ecosystem impacts of intense hurricanes and responses of coastal ecosystems to hurricane disturbances (Horvitz et al. 1998, Stokstad 2005, Day et al. 2007, Chapman et al. 2008, Kupfer et al. 2008, Middleton 2009, Wang and Xu 2009). Kupfer et al. (2008) reported disproportionately high forest damage in the southeastern Mississippi landscape following the Hurricane Katrina. Middleton (2009) evaluated tree seedling regeneration patterns in *Taxodium distichum* swamps in coastal Louisiana and Mississippi following Hurricane Katrina and Rita and reported high mortality of many tree seedlings, but also reported increased seedling regeneration, including the invasive *Triadica sebifera* due to the higher light availability from canopy damage. Chapman et al. (2008) reported a significant increase in regeneration of the invasive *Triadica sebifera* along Louisiana's Pearl River basin following Hurricane Katrina. A widespread mortality of some of the flood intolerant native species and increased regeneration of invasive *Triadica sebifera* has recently been reported in leveed bottomland hardwood forests in Louisiana after the Katrina storm surge (Howard 2012) . However very little information exists from which to determine the responses and recovery potential of invasive and co-occurring native species from low-lying coastal areas under changing environmental conditions (e.g., increased salinity from sea level rise and storm surges, and light availability after canopy wind throw). Therefore, predictions of coastal community responses and their reassembly under changed environmental conditions will require an improved understanding of the responses of native and invasive species to elevated salinity, canopy damage, and possible belowground effects to plant species to changing conditions and environmental stresses (e.g., salinity).

Salinity and Plants Growth-

Salinity is an important physical parameter that adversely affects plant performances (Bernstein 1975; Kozlowski 1997). Salinity limits germination (Ungar 1978; Woodell 1985), vegetative, and reproductive growth of plants by altering ionic and osmotic balance within the plants and their rhizosphere (Kozlowski 1997 and references therein). It has been widely reported that salinity substantially inhibits seed germination of nonhalophytes by disrupting osmotic and physiological processes (e.g., imbibitions) (Ungar 1978) and increasing toxicity to the embryo due to excessive Na^+ and Cl^- content (Zekri 1993). Increased Na^+ and Cl^- reduce vegetative growth in plants by suppressing leaf initiation and by blocking the uptake and transport of water and essential nutrients such as K^+ (Kozlowski 1997) and Nitrogen (Drake and Ungar 1989). High concentrations of Na^+ and Cl^- content in plant tissue cause mottle, burn, and chlorosis in leaves followed by early leaves abscission. Woody plants are more susceptible to injuries brought on by high concentrations of Na^+ and Cl^- (Bernstein 1975). Reduced leaf development and early abscission influences several physiological (e.g., photosynthesis and transpiration), reproductive (e.g., flowering, fruit development, seed production etc.) processes, and fitness in plants (Bernstein 1975; Goldstein et al. 1996; Kozlowski 1997).

To maintain growth and reproduction within saline environments such as low-lying coastal areas, plants must tolerate varying levels of salinity. Some plant species, such as halophytes (e.g., Chenopodiaceae) have the ability to withstand some degree of salinity through osmotic adjustment (compartmentation of Na^+ and Cl^- into vacuoles) and selective uptake of K^+, in order to maintain their productivity (Debez et al. 2008; Flowers and Colmer 2008). Some other plant species, including nonhalophytes, have the ability to tolerate salt through excretion (Kozlowski 1997). Plants species that have the ability to cope with salty environments occupy the most seaward habitats (salt marshes), while salt intolerant plants occupy less saline freshwater marshes and forested habitats, thus creating species zonation along the coastal transitions (Crain et al. 2004).

Effects of Hurricane and Sea Level Rise on Soil Communities-

Besides the visible aboveground effects, hurricane disturbances and sea level rise may have influence on belowground communities and differentially affect plant species and belowground associations (e.g., mycorrhizal-plant mutualisms). Hurricane disturbances can have both positive and negative effects on arbuscular mycorrhizae (AM) fungal communities (Vargas et al. 2010). Vargas et al. (2010) observed nearly a 50% reduction in AM sporulation and a significant increase in AM root colonization after intense hurricane Wilma in the northeast Yucatan Peninsula, Mexico. However, increased salinity due to storm surge and sea level rise may decrease AM root colonization (Juniper and Abbott 1993, Tavares et al. 2012). Decrease AM root colonization may produce negative effects on plant-VAM mutualisms, which is necessary for some plant species to cope with salinity stress. Studies have shown that mycorrhizal fungi can ameliorate stress, such as salt stress, to the host plants by helping in nutrient acquisition (Giri et al. 2007, Daei et al. 2009, Evelin et al. 2009). It was found that the ectomycorrhizal fungus *Scleroderma bermudense* alleviates salt stress in seagrape seedlings by increasing phosphorus (P) and potassium (K) absorption (Bandou et al. 2006). Disturbances influence colonization and species composition of fungal communities and alter the mutualistic association between host plants and mycorrhizae by altering the species of ectomycorrhizal fungi present on plants' roots (Jones et al. 2003). Mycorrhizae formation depends on the soil environment (e.g., soil moisture, temperature, and nutrient availability), host physiology, and soil microorganisms (Entry et al. 2002), which can be disrupted by damaging the hyphal network (Jasper et al. 1989, Hagerman et al. 1999). With the numerous possible effects of climate change, such as increased greenhouse gases, higher temperatures, altered precipitation patterns, increased fires, global sea level rise, shifts in species ranges, and compositional changes, the complex interactions between plants and mycorrhizal fungi may be severely disrupted, leading to disruption of ecosystem functioning (Devall and Parresol 1998).

Climate Change, Shifts in Disturbance Regimes, and Plant Invasions-

Biological invasion is an important component of anthropogenic global change (Vitousek et al. 1997), with effects on the loss of global biodiversity and declines in ecosystem services (Pimentel et al. 2000, Pimentel et al. 2005). The other drivers of global change - such as increased atmospheric CO_2, increased temperature, changing precipitation patterns, land use, fragmentation, and shifts in disturbance regimes – influence resource availability and community dynamics, potentially interact with biological invasions and increase the susceptibility of invasions (Dukes and Mooney 1999, Thuiller et al. 2007, Hellmann et al. 2008, Bradley et al. 2010). Disturbance can create novel empty spaces by generating new resource opportunities, and reducing competition and enemies (Mack et al. 2000). These events may provide opportunities for the spread of exotic alien plants with high rates of fecundity and dispersal (Thuiller et al. 2007). Recently, it was proposed that predicted increases in extreme climatic events (e.g., hurricanes, floods, wind-storms, and drought) will increase the rate of species introductions in various regions (Diez et al. 2012). Studies have demonstrated that hurricane wind disturbances enhance alien plant invasion by increasing light and nutrient resources in affected areas (Bellingham et al. 2005, Chapman et al. 2008, Lynch et al. 2009). Bellingham et al. (2005) recorded a significant increase in total tree density of the invasive *Pittosporum undulatum* in montane rain forests in Jamaica within 6 years following Hurricane Gilbert. A similar trend was also recorded for several alien tree and liana species that invaded subtropical hardwood forests in Florida after hurricane disturbances (Horvitz et al. 1998, Lynch et al. 2009). Furthermore, climate change driven changes in stream flow is predicted to favor stress tolerant invasive species in riverine ecosystems of the western United States (Perry et al. 2012). Increasing global trade (e.g., horticulture) and human movements likely promote species movement into new localities, and their subsequent reproduction is expected under altered site conditions from climate change (Thuiller et al. 2007, Bradley et al. 2012).

However, climate change driven changes in environmental conditions may not always increases the performances of exotic invasive species. Recently, it was suggested that climate change may reduce the habitat suitability and performance of invasive species in some places (Bradley et al. 2009, Bradley and Wilcove 2009). Using bioclimatic envelope models, Bradley et al. (2009) showed a range contraction of three of the five highly invasive plant species under changing climate in the Western US. Paudel et al. (*in prep.*) reported decreased performance of the highly invasive floating aquatic plants, *Salvinia molesta* and *S. minima*, under experimentally elevated salinities that mimic sea level rise and storm surge events for coastal swamp forests of the southeastern US. Given this uncertainty, additional research is needed to examine how native and invasive species will respond to predicted changes in environmental conditions due to climate change.

Role of Mycorrhizal Fungi on Stress Tolerance in Plants and Invasions-

Vesicular arbuscular mycorrhizal (VAM) fungi are ubiquitous in terrestrial (Smith and Read 1997) and wetland (Rickerl et al. 1994) ecosystems. VAM fungi form symbiotic associations with plants and could potentially influence ecosystem processes (Wardle 2002, Klironomos et al. 2011). VAM fungi are important in mineral nutrient uptake, particularly the phosphorus which is not readily available for plants (George et al. 1995, Smith and Read 1997).

Increased nutrition uptake may increase fitness in plants (Barea et al. 2002). Another potential benefit of VAM association is improved stress tolerance of host plants (Giri et al. 2007, Abbaspour et al. 2012). In a glasshouse experiment, Abbaspour et al. (2012) reported improved drought tolerance in mycorrhizal *Pistacia vera* vs. non-mycorrhizal seedlings due to enhanced phosphorus uptake and other physiological responses. Furthermore, Giri et al. (2007) found improved salinity tolerance in mycorrhizal seedling of *Acacia nilotica* seedlings related to increased phosphorus uptake.

Global change and extreme climate events are likely to drive changes in local microclimates, increased abiotic stresses, such as water and temperature stress (Allen et al. 2010), salinity and flooding stress (Williams et al. 1999a), and produce significant effects in plant communities. In a recent review paper Allen et al. (2010) reported a significant increase in tree mortality around the world from drought and heat stress. It was also found that the increased salinity can have significant negative effects on native and invasive plant species in coastal areas (Kuhn and Zedler 1997). Given the expected widespread effects of extreme events, positive benefits from VAM to host plants, which potentially increase stress tolerance, will be significant in responding to environmental stresses. Invasive species occupying newly disturbed habitats are capable of tolerating environmental stress better than some of the native species (Richards et al. 2008, Walls 2010). Additionally, invasive plant species may receive stronger positive feedback through VAM mutualism (Marler et al. 1999a, Callaway and Ridenour 2004, Nijjer et al. 2008) and acquired higher VAM colonization compared to native species (Marler et al. 1999b, Nijjer et al. 2008). Higher VAM colonization and stronger positive feedbacks may provide invasive plant species a competitive advantage over native counterparts, with enhanced tolerance to environmental stresses.

Plant Species Responses to Environmental Changes-

With the predicted increase in extreme climate events (e.g., hurricanes, wind storms, drought, flooding, etc.) (Meehl et al. 2007, Vermeer and Rahmstorf 2009, Mousavi et al. 2011), the populations of species and structure of communities will be affected in many geographical regions (Thuiller 2007). There is no doubt that several plants and animal species are already responding to recent climate change (Parmesan et al. 1999, Parmesan 2006); the responses however, differ between species and functional groups (Klanderud 2008, Sheppard et al. 2012). Greater understanding of species responses to climate extremes and altered environmental conditions is therefore needed to predict structure and community assemblages and futuristic conservation priorities in order to minimize potential biodiversity loss due to invasions. Parmesan et al. (1999) reported poleward shifts of non-migratory butterfly species attributed to regional climate warming. Increased distribution ranges of dominant plant species along elevation gradients due to regional warming have been reported from Southern California's Mountains (Kelly and Goulden 2008). Furthermore, 3-fold increases in woody shrub density in the Sonora Desert of the southeastern US have been attributed to regional climate change (Brown et al. 1997). Further, plant species that responded positively to climate change by advancing their phenology (e.g., early bud break, and flowering) were more likely to increase their performance (increase percent cover, biomass, individual growth, and flower numbers) (Cleland et al. 2012).

Recent studies provide evidence for the positive responses of exotic invasive plant species to climate change (Cleverly et al. 1997, Ziska 2003, Dukes et al. 2011), but see Bradley et al. (2009) and Sheppard et al. (2012). Cleverly et al. (1997) studied drought tolerance and invasive capacity of *Tamarix ramosissima* in a Mojave Desert floodplain and they found that *T.*

ramosissima outperformed other three co-occurring native species under extended drought conditions. They concluded that the greater dominance of *T. ramosissima* will be possible under extreme drought events that are predicted for the region. Smith et al. (2000) found a higher aboveground biomass and seed production of annual invasive grass *Bromus madritensis* than native annual species under elevated CO_2 conditions in an arid ecosystem of western US. Ziska (2003) grew six invasive plant species, which are widely considered to be among the most invasive species in the continental United States, at three different levels of CO_2 concentrations, corresponding to past, current, and future (IPCC 2001 projections) concentrations. She reported a significant increase in performance of those species (e.g., plant biomass) with elevated CO_2 concentration. More recently, He et al. (2012) conducted an experiment and compared performance of the invasive *Eupatorium adenophorum* and a native congener *E. chinense* under artificial warming. They found that the invasive *Eupatorium* outperformed native *Eupatorium* in all ecological measurements they conducted, suggesting that a warmer world would likely be friendlier to some exotic invasives than native species. Dukes et al. (2011) reported increased growth of an invasive *Centaurea solstitialis* compared to resident species in response to elevated CO_2 in California, USA.

In general, invasive species might be expected to respond positively to climate change because of their greater physiological and morphological plasticity to tolerate a wide range of climatic and environmental conditions compared to native and non-invasive species (Daehler 2003, Richards et al. 2006, Davidson et al. 2011). It has also been suggested that the species with greater phenotypic plasticity may have greater capacity to tolerate changing environmental conditions (Ghalambor et al. 2007). When invasive species demonstrate greater phenotypic plasticity than non-invasive species (Richards et al. 2006, Davidson et al. 2011), such plasticity also increases their fitness (Davidson et al. 2011). There are several other reasons why invasive species respond positively to environmental changes. Exotic invasive species in new locations are already released from their natural enemies and pathogens (above- and/or below-ground) (Keane and Crawley 2002). In addition, they may exploit additional resource opportunities and devote more resources to growth and reproduction (Daehler 2003) as native species are struggling to cope with natural enemies, increased competition, and environmental changes. Finally, invasive species that receive stronger positive feedbacks than natives, i.e., enhanced mutualism (Reinhart and Callaway 2006), from mycorrhizal fungi may gain a competitive advantage if stress tolerance is improved with of increased uptake of limited nutrients (Reinhart and Callaway 2006).

The small number of available studies that have examined the response of native and invasive species to altered environmental conditions was predominantly focused on these species' growth responses to elevated CO_2, changes in precipitation gradient, and temperature (Dukes 2002, Ziska 2003, He et al. 2012). Limited data are available, however, on potential responses of invasive and co-occurring native species to other environmental changes triggered by accelerated sea level rise and intensification of tropical storms along low-lying coastal ecosystems. Conner et al (1997) investigated the effects of saltwater flooding on growth and survival of an invasive *Triadica sebifera* and three co-occurring native species of forested wetlands of coastal Louisiana and demonstrated that species were differentially affected by flood and salt stress. From these results, they concluded that the differential stress tolerance of native and invasive species may cause shifts in species composition. More recently, Howard (2012) found improved regeneration and dominance of the invasive *Triadica sebifera* after flooding from Hurricane Katrina in 2005 in coastal bottomland hardwood forests of Louisiana. He also reported a decreased survival and regeneration of some of the native species (e.g., *Ilex vomitoria*) in this system.

Given the ambiguities regarding the responses of invasive and co-occurring native species to intense hurricane disturbances and sea level rise, further research is needed to examine these species responses under altered environmental conditions in order to fully understand the directions and community dynamics of these low-lying coastal ecosystems in response to environmental changes. To my knowledge, there do not appear to be any previous studies documenting responses of invasive and co-occurring native species, growing in different locations along coastal transitions, to predicted environmental changes (e.g., elevated salinity, increase storm water flooding, canopy damage etc.). Since, a large portion of the present day human population relies on coastal ecosystems for numerous goods and services (Martinez et al. 2007), a better understanding of plant species' responses to environmental changes can lead to appropriate management goals and planning for the future for these critical marine – terrestrial ecotonal ecosystems already experiencing effects of climate change.

Coastal Ecosystems of the Southeastern United States and Triadica sebifera Invasion-

In the southeastern United States (US), one of the highly invasive plants is *Triadica sebifera*, commonly called Chinese tallow. This species occurs in great abundance in many coastal ecosystems of the southeastern US, especially in low and flat lands, abandoned fields, pastures, degraded areas, roadways, and forests. *Triadica sebifera* is a small- to medium-sized subtropical, deciduous, monoecious tree in the family Euphorbiaceae and was introduced into the US for ornamental purposes in the late 18th century from east Asia (Randall and Marinelli 1996), and has been naturalized across the southeastern USA (Jubinsky and Anderson 1996, Bruce et al.

1997). *Triadica sebifera* is highly shade tolerant and capable of invading different habitats ranging from swampy areas, flooded areas, floating marsh, and moderate level salt water to upland forested areas (Jones and McLeod 1989, Jubinsky and Anderson 1996, Conner et al. 1997, Battaglia et al. 2009). This species has tremendous reproductive potential and can reach in reproductive stage within its first year (*pers*. *comm*. with JB Grace), and a mature tree annually produces up to 100,000 seeds (Jubinsky and Anderson 1996). Seeds are readily dispersed by native migratory birds (Renne et al. 2002). This is an aggressive invader and once established, it has the capacity to reduce native diversity by displacing native species (Bruce et al. 1995).

There is ample research that describes the *Triadica sebifera* invasion throughout the southeastern US (Bruce 1993, Bruce et al. 1995, Rogers and Siemann 2003, Rogers et al. 2003, Battaglia et al. 2009, Gan et al. 2009). Furthermore, increased recruitment of this species after hurricane wind disturbances was recently been reported from coastal Louisiana (Chapman et al. 2008). Recently, it was also projected that, under increased temperature and precipitation, *Triadica sebifera* will expand its population further inland beyond its current distribution range (Pattison and Mack 2008). Noticeably missing, however, are the field observational studies that describe the effects of two important components of climate change, sea level rise and increased intensity of hurricanes on distribution of *Triadica sebifera* in conjunction with co-occurring native species along the coastal transitions. Also, limited information is available on the environmental (e.g., salinity, shade, and light) tolerance of this species along the transitions. Additionally, we are lacking experimental studies that test the responses of *Triadica sebifera* and co-occurring native species to predicted environmental changes (e.g., elevated salinity, storm sure, and tree canopy damage) from 21st century sea level rise and hurricane events. Understanding responses of *Triadica sebifera* and co-occurring native species to future environmental changes can provide important insights into this aggressive invaders' invasive capacity in the forests of the Southeastern US already experiencing the effects of climate change. For the purpose of this study, I used *Triadica sebifera* as a model invasive species and selected three native species; *Baccharis halimifolia, Ilex vomitoria*, and *Morella cerifera*, that are common in the Atlantic and Gulf Coastal Plains and commonly share similar habitats with *Triadica sebifera*. A general description of these species is given in Chapter 2.

This dissertation thus aims to advance understanding of the *Triadica sebifera* invasion under disturbed and changing environmental conditions across forests of the southeastern US, and may inform land managers for developing future management plans in order to curb further expansion of this aggressive invader. In my first data chapter, I assessed the distribution of *Triadica sebifera* and selected native species along a typical coastal transition at Grand Bay National Estuarine Research Reserve (GBNERR), coastal Mississippi (for detailed description of the study area, see chapter 2). I tested species distribution range along the coastal transition by relating species' probability of occurrence to various environmental variables. In the Chapter 3, by using growth chamber and greenhouse experiments, I tested the effects of elevated salinity predicted with sea level rise on initial recruitment (germination) of the invasive *Triadica sebifera* and two native species, *Baccharis halimifolia* and *Morella cerifera* in soils representing different habitats along the coastal transition. In Chapter 4, I manipulated two important variables of intense hurricane disturbance (i.e., canopy damage and storm surge) and monitored growth response of target species (the invasive *Triadica sebifera* and two native *Baccharis halimifolia* and *Morella cerifera*) for two growing seasons in order to test potential effects of predicted intense hurricane disturbances. Finally (chapter 5), I studied mycorrhizae dependency of the invasive *Triadica sebifera* and two native species, *Baccharis halimifolia* and *Morella cerifera* on their establishment and growth and the role of mycorrhizae in stress tolerance and survival under increased salinity from hurricane generated storm surge.

CHAPTER 2

ABIOTIC CONSTRAINTS ON JUVENILE OCCURRENCE OF INVASIVE AND NATIVE PLANT SPECIES: IMPLICATIONS FOR FUTURE DISTRIBUTIONS WITH CLIMATE CHANGE

ABSTRACT

Understanding relationships between natural distribution patterns of species and their surrounding environment provides a basis for forecasting how species will respond to future environmental changes. However, the relative importance of surrounding physical environments in determining the invasibility of habitats to exotic species is not well understood, particularly across the range of habitats in coastal wetlands. The goal of this study was to assess the probability of occurrence of juveniles of invasive species *Triadica sebifera* and three native species, *Baccharis halimifolia*, *Ilex vomitoria*, and *Morella cerifera* with respect to environmental factors, and then use the occurrence models to predict habitat invasibility under future environmental changes. The occurrence of *I. vomitoria* and *M. cerifera* and the invasive *T. sebifera* sharply decreased with increased salinity. In contrast, *B. halimifolia* occurrence did not decrease until salinity happened to be really high. The occurrence of *T. sebifera* was significantly associated with landscape structures. Given the observed pattern of native and invasive species occurrences, I predict that future increases in salinity will negatively impact *I. vomitoria*, *M. cerifera*, and *T. sebifera*, leading to range contraction of these species away from the coast. However, I suggest the natural (e.g., canopy damage) and anthropogenic disturbances (e.g., construction, fragmentation, recreational activities etc.) may offset the effects of salinity stress by increasing resources, propagule dispersal, and possibly increase community vulnerability to *T.*

sebifera invasion. If the current occurrence pattern of *T. sebifera* juveniles is inferred as its invasive potential, predicted shifts in disturbance regimes are likely to increase the dominance of this invader in already invaded areas and promote new invasions in forestlands of the southeastern USA.

INTRODUCTION

Disturbances are key processes that can drive changes in the community structure by creating regeneration opportunities for disturbance tolerant species (Hobbs 1989, Hobbs and Huenneke 1992). Climate change may alter the frequency and intensity of disturbances (Dale et al. 2001, IPCC 2007a) and increase ecosystem vulnerability to the proliferation of invasive species (Dukes and Mooney 1999, Burgiel and Muir 2010), particularly those with high rates of fecundity and dispersal (Thuiller et al. 2007). Furthermore, disturbances can create novel environments by generating new resource opportunities, enemy reduction, and expansion of empty spaces (Mack et al. 2000). Once established, alien invaders can outcompete native species (Bruce et al. 1995, Daehler 2003) and cause a significant decrease in ecosystem services (Williamson 1996, Zavaleta 2000).

Low-lying coastal wetland ecosystems along the northern Gulf of Mexico and Atlantic coasts are highly vulnerable to sea level rise and tropical storms (IPCC 2007a, Titus et al. 2009). Intense tropical storms produce strong winds that cause substantial damage to forest canopy by toppling trees, snapping branches, and defoliation (Bellingham et al. 1995, Michener et al. 1997, Chambers et al. 2007). Windstorm damage in forest stands increase canopy gaps, which enhance light availability to lower strata, and greatly change the micro-environmental conditions (Bellingham et al. 1996, Battaglia et al. 1999). Increased light availability, altered micro-environmental conditions (e.g., temperature and moisture), and new resource opportunities likely enhance the proliferation of exotic invasive species (Davis et al. 2000, Parendes and Jones 2000, Laurance and Curran 2008). In addition, intense tropical storms deliver saltwater farther inland than usual through storm surges and salt spray (Moorhead and Brinson 1995), which may have stronger negative impacts on more inland plant communities that are not adapted to saline conditions (Tate and Battaglia 2012). Storm surges also alter soil substrates through erosion and sedimentation, which may drive changes in the distributions of coastal species (Williams et al. 1999b). Predicted acceleration of sea level rise (Meier et al. 2007, Rahmstorf 2007) and increased intensity of hurricanes (Bender et al. 2010) will likely exacerbate the environmental changes, leading to long-term shifts in coastal plant assemblages (Scavia et al. 2002, Hopkinson et al. 2008).

Coastal and estuarine ecosystems are one of the highly invaded ecosystems, assisted in part by anthropogenic activities that transport invasive species, and large scale natural and anthropogenic disturbances (Cohen and Carlton 1998). For instance, urbanization and anthropogenic nutrient have been shown to increase *Phragmites australis* invasion on New England salt marshes (Silliman and Bertness 2004). Following natural hurricane disturbance (hurricane Andrew in 1992), non-native vines expanded rapidely after wind-driven canopy loss and inhibited regeneration of native species in Florida (Horvitz et al. 1999). In Pearl River basin in Louisiana, an improved regeneration of invasive *Triadica sebifera* (Chapman et al. 2008) and invasive *Rubus* sp. (Brown et al. 2011) was observed following canopy loss from Hurricane Katrina in 2005. Since its introduction, one of the highly aggressive invasive species, *Triadica sebifera* (L.), has been spreading rapidly in various coastal plant communities, such as coastal prairie (Bruce et al. 1995), floating marsh (Battaglia et al. 2009), and forested wetlands (Gan et al. 2009, Wang et al. 2011). Widespread invasion of *Triadica sebifera* is producing damaging impacts on native plant (Bruce et al. 1997) and fauna communities (Pattison and Mack 2008). Although, it has already been more than a decade since *Triadica sebifera* was listed among the “least wanted intruders” in United States (Flack and Furlow 1996), studies that link juvenile distribution and habitat preferences are rare.

The physical environmental features play an important role in community invasibility (Holway et al. 2002, Shea and Chesson 2002). It has been suggested that extreme physical environments may act as a barrier on invasions (Richardson et al. 2000b). Studies have suggested that low nutrient availability and drought often reduce habitat invasibility (Alpert et al. 2000). Conversely, extreme disturbance events such as windstorms and floods may increase prevalence of invasive species by increasing dispersal and creating resource-rich novel environmental conditions (Diez et al. 2012). These conditions may favor the established and dominance of exotic invasives, which in some cases may outcompete resident species (Bruce et al. 1995; Mack et al. 2000; Bradley et al. 2010). In addition, certain landscape structures (roads, water bodies, power lines, abandoned fields, and recreation sites) may play a critical role in the spread and establishment of invasive species (Bradley and Mustard 2006; von der Lippe and Kowarik 2007; Flory and Clay 2009; Gan et al. 2009). These landscape structures constitute the major dispersal corridors for invasive plants into the natural areas. Roads and water bodies serve the obstruction-free paths for seed dispersal by birds and water (Parendes and Jones 2000) while power lines and forest edges may serve as important perch and nesting sites for birds.

Some invasive species that exhibit phenotypic and plasticity and may overcome various physical barriers and ultimately prevail (Richards et al. 2006). The invasibility of habitats and governing factors (i.e., either promoters or barriers) that influence the invasive *Triadica sebifera,* or other invasive species, in early life stages needs to be examined particularly in regions where the abiotic environment is changing on a regular basis due to the effects of climate change, such as low-lying coastal areas of the southeastern USA. However, effects of environmental factors, natural and anthropogenic disturbances and landscape structures on the juvenile occurrence of *T. sebifera* and co-occurring native species in the coastal transition ecosystems of the southeastern USA have not been evaluated.

The seedling regeneration stage is important for plant establishment (Grubb 1977) due to their greater sensitivity to environmental stress (Harper 1977). Once the seedlings or juveniles survive under particular environments, they grow and gradually gain adult traits typical of higher stress tolerance (Rey and Alcantara 2000). The successful juvenile establishment often shape the spatial dynamics of recruitment of plant species and may determine the succession and future trajectories of communities (Rey and Alcantara 2000). Thus, relating species' distributions to environmental predictors is important for determining the ranges and suitable habitats for the persistence of species in the landscape (Araujo and Williams 2000). This information can also be used to identify potential sites at risk of new invasions in order to adopt appropriate monitoring and control measures (Thuiller et al. 2005). Here, I examined the role of physical factors (light, salinity, and soil characters) and landscape structures (roads, trials, power lines, abandoned fields, human settlements, water bodies etc.) to assess probability of juveniles occurrence of the aggressive invader, *Triadica sebifera,* and common co-occurring native species in a typical coastal transition of southeastern USA. By examining invasive and native species distributions to surrounding environmental conditions, we can learn more about community susceptibility to invasions and predict future community assembly in the face of changing environmental conditions.

The objectives in this study were (i) to use the field distributions of juveniles of the target species to estimate their probability of occurrence with respect to various environmental factors and landscape features and (ii) to use occurrence models to predict the potential effects of predicted shifts of disturbance regimes (e.g., increase intensity of tropical storms, and sea level rise) on future distribution patterns of these species in the coastal ecosystems of southeastern USA.

MATERIALS AND METHODS

Study Area-

The study was carried out in the Grand Bay National Estuarine Research Reserve (GBNERR) in coastal Mississippi on the northern coast of the Gulf of Mexico. The reserve, which was established in 1999, is located in southeast Mississippi between Pascagoula and the Alabama state line (30°21′23″ N and 88°27′46″ West, Figure 2.1), and occupies 7,446 hectares of land. The warm and humid-temperate climate of the region is characterized by long growing seasons (Otvos 2007).

Coastal Mississippi has been frequently impacted by hurricane disturbances (Peterson et al. 2007). Intense hurricanes disturb the existing vegetation and soils and deliver enormous storm surges that inundate vast coastal areas. For example, Hurricane Katrina inundated the entire portion of GBNERR with a storm surge that was up to 8 meter high, and the storm surge remained for several days (Peterson et al. 2007). High winds from Katrina produced canopy gaps in interior forests. The GBNERR lies within the gently sloping lower Gulf Coastal Plain and forms one of the most biologically diverse and productive estuarine ecosystems on the Gulf of Mexico (Hilbert 2006, NOAA 2007). The reserve supports various coastal wetland plant communities shaped by the environments of the northern Gulf of Mexico (Peterson et al. 2007). Tidal activities, hydrology, elevation, and salinity are the major environmental factors that determine the different plant communities in these northern Gulf Coast ecosystems (Peterson et al. 2007). Tidal communities of salt and brackish marshes dominate at the seaward end of the gradient and slowly grade into freshwater marshes, maritime forests, cypress swamps, mixed hardwood forests, wet pine flatwoods, pitcher plant bogs, and pine savannahs toward the landward end of the transition.

Within the reserve a permanent transect (11.3 km long) was established in the reserve in 2007 to study the effect of natural disturbances (e.g., tropical storms and sea level rise) and altered environmental conditions (e.g., hydrology and salinity) on estuarine-upland transitions (Figure 2.1). The transect, oriented north-south, starts at the marine-terrestrial interface along the Gulf Coast and ends in pine savannah; it was used as a reference for this study. The transect encompasses a typical coastal transition supporting various aforementioned plant communities with gradients of elevation (Figure 2.2), salinity measured as soil water conductivity (Figure 2.3), flooding, and soil properties.

Target Species-

I chose three native species, *Baccharis halimifolia* L. (Asteraceae), *Ilex vomitoria* Aiton (Aquifoliaceae), *Morella cerifera* L. (Small) (Myricaceae), and the invasive species *Triadica sebifera* L. (Small) (Euphorbiaceae) for this study. All of these species co-occur in various wetland habitats along the northern Gulf of Mexico coast (Bruce et al. 1995, Wall and Darwin 1999, Battaglia et al. 2009).

Baccharis halimifolia-

Baccharis halimifolia is a native, multi-branching deciduous, perennial understory shrub which occurs along the Atlantic and Gulf coasts of North America. This species is most commonly found on the southeastern coastal plains (Duncan et al. 1957). It grows in moist soil having high organic matter, such as open forests, beaches, swamps, wet prairie, and fresh water marshes (Penfound and Hathaway 1938, Duncan et al. 1957). Because of high salinity tolerance (Moon and Stiling 2004), it also occurs along the fringe of brackish marshes (Penfound and Hathaway 1938, Duncan et al. 1957). It is shade intolerant species and produces wind dispersed seeds.

Ilex vomitoria-

Ilex vomitoria is an evergreen shrub native to coastal forests along the Gulf and Atlantic Coastal Plains of the United States. This species is found in maritime forests, upland woodland, wet-woodlands, floodplains, and non-tidal forested wetlands with well-drained sandy soils (Godfrey 1988). *Ilex vomitoria* fruits are a good source of food for many native and migratory birds and fruits remain on the tree until late fall to early spring. The species grows well in sunny sites as well as under shaded conditions (Chambless and Nixon 1975).

Morella cerifera-

Morella cerifera is an evergreen nitrogen-fixing shrub native to coastal plains of the southeastern US, which is said to be shade intolerant and moderately salt tolerant (Duncan and Duncan 1987, Tolliver et al. 1997). *Morella cerifera* is an early successional species in southeastern coastal regions (Young et al. 1995). In floating freshwater marshes of coastal Louisiana, this species has been shown to have facilitative effects on initial recruitment of invasive *T. sebifera* (Battaglia et al. 2009).

Triadica sebifera-

Triadica sebifera is a fast growing deciduous tree that has invaded many southeastern coastal plant communities. *Triadica sebifera* exhibits many traits associated with successful invaders, such as fast growth rate (Bruce et al. 1997), effective seed dispersal by birds (Renne et al. 2000, Renne et al. 2002), and a wide range of tolerance to environmental conditions (e.g. soil salinity, flooding, and shade) (Conner and Askew 1993, Jubinsky and Anderson 1996, Barrilleaux and Grace 2000). It has a high reproductive potential and may reach reproductive age within its first year (*pers. comm.* with JB Grace). The species appears to be an aggressive invader and causes large-scale ecosystem modification by displacing native plant species and forming monospecific *T. sebifera* woodland (Bruce et al. 1995, Jubinsky and Anderson 1996).

Sampling Method-

An observational study was carried out along the entire transect (except a section from7300 m to 8200 m due to private landowner inholdings) in May-August, 2008. Preliminary field observation showed that no woody plant species were present in the first 4.5 km section of the transect. Thus, only soil samples and canopy photo were taken in that section at 100 m intervals. For the remaining portion of the transect, which was predominantly forested, presence or absence of focal species and quantification of environmental variables were assessed at 25 m intervals. This differential sampling was implemented due to a marked difference in vegetation composition.

We established a random point, 12 m away from each reference point, on the transect in either an east or west direction randomly. A juvenile (≤ 1.5 m height) of each target species, when present, was located in the opposite direction, between 1- 12 m of the reference point. When more than one juvenile of focal species was encountered, we identified the nearest one from the reference point. At each random point, we recorded the presence/absence of juveniles of the focal species and quantified the canopy openness by taking hemispherical photographs at a height of 1.5 m immediately above the point. Similarly, at each juvenile location, we recorded the presence of the juvenile and quantified canopy openness over each juvenile at a height of 1.5 m. Soil samples of the top 30 cm were also taken at each random point and juvenile locations using a 2 cm diameter soil corer. Additionally, we recorded landscape structures (roads, trails, railroads, power lines, water bodies, and recreational sites) during the survey. We assigned the presence of the aforementioned landscape structures in each random point and juvenile locations within 50 m of the identified structures. During data analysis, we combined all the identified structures as a single landscape variable.

Soil Analysis and Canopy Openness Determination-

Soil samples were used to measure soil salinity, soil texture, and carbon/nitrogen. All the soil samples were air dried, pulverized and sieved with a mesh size of 2.0 mm and oven-dried at 55°C. Soil salinity was determined by measuring the electrical conductivity (EC) (electrical conductivity is directly proportional to salinity) using a conductivity meter (EcoSense EC300; YSI Inc., Yellow Springs, Ohio, USA) following slight modification to the procedure describe in Methods of Soil Analysis (Rhoades 1996). Soil texture (contents of sand, silt, and clay) was determined by using the hydrometer method (Sheldrick and Wang 1993). A small subsample (7g) of soil was dried at 55°C, ball-milled to a fine powder to determine % C and % N using a Thermo Flash 1112 CN Analyzer (Thermo Corp, New Jersey). Canopy photographs were used to estimate canopy openness, a function of canopy disturbance. I used a digital camera and fisheye lens (Nikon Coolpix 4500) to take a photograph of the canopy from 1.5 m above the ground. Each image was imported into Gap Light Analyzer (GLA), a software program for quantifying canopy openness (Frazer et al. 1999). These data were used to determine the probability of occurrence of the target species along the coastal transition at Grand Bay.

Statistical Analysis-

Probability of occurrence of invasive *T. sebifera* and the native plant species at particular sampling points, based on that point's characteristics, was evaluated by using logistic regression (PROC GENMOD in SAS) (Allison 1999). Since the data are binary (presence/absence), I used generalized linear models (GLM) for all logistic regression analyses. Predictor variables for inclusion in the logistic model were: distance to coast, soil water conductivity, percent canopy openness, percent sand, percent silt, percent clay of soil, soil C/N, and landscape features. Before the logistic models were developed with the measured environmental variables, I evaluated the probability of occurrence of target species in relation to distance from the coast. Logistic regression models were constructed for each variable separately and the two variables (soil water conductivity and canopy openness) combined. In the process of logistic model development, I first fit target species presence to each predictor variable itself and then combined two factors (soil water conductivity and canopy openness) into more complex models. I compared these combined models to each of the simpler one-factor models. Finally, I added an interaction term (soil water conductivity × canopy openness) to determine the fit of the more complex model. If the fit of the model was significantly improved (a significant reduction in the change in deviance) compared to single-factor and combined models, I retained the interaction terms into the final model. A simple single factor (i.e., soil water conductivity) model was required to explain the probability of occurrences for *I. vomitoria* but interactions (soil water conductivity × canopy openness) models were required to improve the predictive ability of the models for *B. halimifolia*, *M. cerifera*, and *T. sebifera* (Table 2.1). Since some of the logistic models are based on a logit transformation, the output was back-transformed to generate separate response curves for each species. A chi-square test was performed to evaluate the association between the probability of occurrence of species and landscape structures by using Contingency Table Analysis. Logistic regression models and Contingency Table Analyses were run with SAS version 9.2 (SAS 2008), and model significance of the variable was compared to the appropriate critical χ^2 value at the $\alpha = 0.05$ level.

RESULTS

Based on the measurements taken at random and tree points along the transect, a wide range of soil water conductivity, canopy openness, soil texture and carbon nitrogen ratios were found. Soil water conductivity ranged from 0.0154 to 18.460 mS/cm. While salinity levels frequently change across the coastal transitions, these measurements were a snapshot of soil water conductivity prior to Hurricanes Gustav and Ike in 2008. Soil water conductivity was higher towards the coast and progressively decreased with distance from the coast (Figure 2.3). The soil texture differed along the transect with soil sand content decreased and clay content increased along the transect from the coast to inland (Figure 2.4). Among the target species, none of the species occurred in the first 4500 m of the transect, which was essentially brackish marsh habitat.

Distance to coast strongly affected the probability of occurrence of species (logistic regression: *B. halimifolia*, DF = 1, Wald $\chi^2 = 10.18$, $P = 0.0014$; *I. vomitoria*, DF = 1, $\chi^2 = 27.92$, $P = < 0.001$; *M. cerifera*, DF = 1, $\chi^2 = 22.45$, $P = < 0.001$; *T. sebifera*, DF = 1, $\chi^2 = 4.09$, $P = 0.043$). None of the species occurred in the first 4500 m of the transect, which was essentially salt and brackish marsh habitat. *Baccharis halimifolia* and *T. sebifera* were absent in most inland portion of the transect (Figure 2.5a and 2.5b, respectively). In contrast, *I. vomitoria* and *M. cerifera* were encountered more frequently throughout the forested portion (i.e., 4.5 to 11.3 km) of the transect (Figure 2.5c and 2.5d, respectively).

Probability of occurrence of *B. halimifolia* was significantly influenced by an interaction effect of soil water conductivity and canopy openness (Table 2.1). Its occurrence increased under some degree of salinity and wide range of canopy openness but decreased with low salinity and high canopy openness (Figure 2.6a). Probability of occurrence of *B. halimifolia* significantly increased with percentage increase of sand and decrease of silt (Figure 2.7a and 2.7b, respectively and Table 2.2). Probability of occurrence of *I. vomitoria* was significantly reduced in areas with high soil salinity but invariant with canopy openness. Measured soil characteristics were not predictive of probability of occurrence of *I. vomitoria* ($P > 0.05$). The probability of occurrence of *M. cerifera* peaked at the lower level of soil water conductivity with low to high levels of canopy openness (Figure 2.6b). The response surface of the species was based on a monotonic response of soil water conductivity and the interaction between soil water conductivity and canopy openness (Figure 2.6b). I did not find significant effects of other environmental factors on the occurrence of *M. cerifera*.

The probability of occurrence of *T. sebifera* showed a monotonic response to soil water conductivity and percent canopy openness, as well as an interaction between these two factors (Table 2.1). The probability of occurrence peaked towards the lower salinity level with low canopy openness (Figure 2.6c). Measured soil characteristics were not predictive of probability of occurrence of *T. sebifera*.

I did not find a significant association between probability of occurrence of *B. halimifolia* and landscape structure, χ^2 (1, N= 320) = 0.05, $P = 0.47$. Similarly, landscape structure was not predictive of probability of occurrence of *M. cerifera*, χ^2 (1, N= 450) = 0.26, $P = 0.60$. In contrast, I found an association between probability of occurrence and landscape structure for both *I. vomitoria* (χ^2 (1, N= 339) = 6.63, $P = 0.01$) and *T. sebifera* (χ^2 (1, N= 332) = 49.17, $P < 0.0001$). Probability of occurrence of both the species was grater with the presence of landscape structure.

DISCUSSION

In this study, I examined the probability of occurrence of the juveniles of invasive *T. sebifera* and native *B. halimifolia*, *I. vomitoria,* and *M. cerifera* in relation to surrounding physical environments along a typical coastal transition in the southeastern USA. I found that juvenile distribution of the invasive *T. sebifera* was patchy, but significantly associated with landscape features. Likewise, *B. halimifolia* distribution was patchy and with higher probabilities of occurrence under wide range of canopy openness and moderate level of salinity. However, juveniles of *I. vomitoria* and *M. cerifera* were found throughout the forested habitats in this study area. *Ilex vomitoria* occurrence was associated with landscape features. The distributional patterns of native and invasive species could be related mainly to spatial variation in soil water conductivity and landscape features including disturbances. These results underline the importance of abiotic factors in controlling plant species distributions along coastal gradients (Crain et al. 2004). Except for *B. halimifolia*, which was present in salt pannes, the other target species had restricted distributions towards the seaward end of the transition. Plant communities at that end are dominated by open marshes which are exposed frequently to tidal flooding and higher salinity. Recurrent flooding with saline water and high salinity likely prevent the establishment of woody species as they have a more narrow tolerance range compared to halophytic grasses and sedges (Kozlowski 1997). Field transplant experiments have also demonstrated that these species were unable to establish in locations seaward of their present distributions (Chapter 5).

Results indicated that *B. halimifolia* occurrence did not decrease until salinity levels reached 4 mS/cm, reflecting its salt tolerance capacity. These findings were not unexpected because *B. halimifolia* is reported to grow and withstand saline conditions in barrier island in Virginia (Young et al. 1994), coastal Florida (Moon and Stiling 2004), and marshes of coastal Mississippi (Eleuterius 1972). Results further demonstrated that *B. halimifolia* occurrence was highest between 4500 m and 6000 m (Figure 2.5a). This section of the transect is closer to the coast and likely to be wetter from tidal activities, suggesting that the probability of occurrence of *B. halimifolia* is associated with wetter habitat as reported previously in the southeastern

(Krischik and Denno 1990) and south-central (Ervin 2009) USA. However, *B. halimifolia* was absent from some of the pristine and nutrient poor habitats, likely due to its preference for more nutrient rich habitats (Sims-Chilton et al. 2010). The occurrence of *I. vomitoria* and *M. cerifera* appears to be significantly constrained by soil water conductivity. These results are consistent with previous findings in which these species were listed as the least salt tolerant in various coastal ecosystems. In an experimental study (chapter 3), I reported a greater negative effect of brackish marsh soils and elevated salinities on germination of *M. cerifera* compared to *B. halimifolia* and *T. sebifera* in coastal Mississippi. On Virginia barrier islands, *M. cerifera* showed a greater sensitivity to salinity compared to co-occurring *B. halimifolia* (Young et al. 1994). Furthermore, *I. vomitoria* and *M. cerifera* species were reportedly growing in upland habitats with low salinity and minimal to no tidal influences (Martin and Mott 1997, Moon and Stiling 2004).

The patchy nature of invasive *T. sebifera* occurrence along the coastal transition in this study provides support for abiotic constraints and propagule limitations (Grubb 1977) impeding juvenile establishment during the early stages of invasion. For any tree species to be able to dominate the forest canopy, it has to pass through important physical filters during its early life stage (e.g., seedlings or juveniles) (Beckage et al. 2000). Decreased probability of juvenile occurrence with increased soil water conductivity suggests that there is an important abiotic constraint operating across the sea-inland environmental gradient dictated by soil salinity that is likely to limit the seaward spread of *T. sebifera* in this study area. Although *T. sebifera* is moderately salt tolerant (Conner 1994), and can handle a modest duration of saltwater flooding than co-occurring native species (Howard 2012), my results suggest that the higher soil salinity, particularly at seaward habitats, likely limits the establishment of this species. These results are consistent with the findings of Barrilleaux and Grace (2000) from a coastal prairie in Texas. They found significantly lower survival of transplanted *T. sebifera* seedlings at the saltier (higher electrical conductivity) end of the gradient.

Some inland habitats (e.g., pine flatwoods and pine savanna) are also not currently invaded by *T. sebifera*, which may reflect insufficient numbers of viable seeds carried to the site (i.e. both dispersal and recruitment limitation) for that species to germinate and establish, or that the conditions are not suitable for seed germination and seedlings survival (for instance, disturbed and moist soil) (Grubb 1977). Most of the sites currently unoccupied by invasive *T. sebifera* are relatively pristine and remain dry for a longer duration. Since *T. sebifera* regeneration and successful seedling establishment are correlated with wet habitats (Neyland and Meyer 1997, Conner et al. 2002, Denslow and Battaglia 2002, Burns et al. 2004), it is possible that dry and less disturbed conditions limit its establishment. Given *T. sebifera*'s ability to germinate in various inland habitats including pine savanna (Pattison and Mack 2008, Chapter 3), I anticipate that propagule limitation is the primary constraint on its invasion in pine savanna. My prediction is consistent with an earlier study that reported *Alliaria petiolata*'s absence, but was able to germinate and establish itself within the experimental plots when seeds were sown into a field, in a hardwood forest in southeastern Ohio, USA (Meekins and McCarthy 2001).

I found evidence that while physical features in the invaded habitats restrict the spread of invasive species in some habitats, natural and anthropogenic disturbances promote the invasions in others (Richardson et al. 2000b). Often these landscape structures, center of anthropogenic activities, are considered as disturbance features and linked to exotic invasion (Bradley and Mustard 2006). Increasing anthropogenic presence in and around roads, power lines, water bodies, and recreation sites for road building, power line emplacement, and recreational activities increase the disturbance in the sites and promote new invasions through increase resource pulse and dispersal (Bradley and Mustard 2006). I found a significant positive relationship between landscape structures and *T. sebifera* occupancy. This is consistent with a previous study that reported a close association between landscape features (water bodies, roads, fire, timber harvest, etc.) and probability of *T. sebifera* occupancy in forestlands of southeastern USA (Gan et al. 2009). In general, at their initial stage of establishment (*T. sebifera* is in the early stage of invasion in this study area (Matlack 2002), invasive species occupy disturbance corridors such as roadsides, power-lines, and human settlements (Theoharides and Dukes 2007). Disturbance not only enhances resource availability (Davis et al. 2000) but also creates many open spaces for species to colonize through easy short distance dispersal in the early stages of invasion (Parendes and Jones 2000, Theoharides and Dukes 2007). For instance, Parendes and Jones (2000) encountered invasive plant species more frequently close to streams and active roads than farther away in the HJ Andrews experimental forest, Oregon, USA, which suggests that the higher resource (light) availability and dispersal via streamflow enhances invasion around roads and water bodies. In recent study (Gan et al. 2009) showed a positive relationship between *T. sebifera* occupancy and water bodies (rivers, streams, bayous etc.) in forestlands of coastal southeastern USA. Increased abundance of invasive plant species with anthropogenic disturbance, and decreased abundance with increasing distance to their source populations have also been reported in the forest on the Cape Peninsula, South Africa (Alston and Richardson 2006). Using logistic models Giorgis and co-authors found the highest probability of occurrence of woody invasive species near established propagule sources, human settlements, and roads in a newly invaded region of central Argentina (Giorgis et al. 2011). Once established, invasive species slowly make their way to more interior and pristine parts of the forest, aided by disturbance (Theoharides and Dukes 2007). In this study area, I observed some well- established *T. sebifera* populations in forest stands whose canopy had been damaged by Hurricane Katrina winds in 2005. Additionally, heavy concentrations of *T. sebifera* in and around ditches and water bodies may be related to easy dispersal of seeds through hydrochory (Bower et al. 2009) and subsequent germination and growth (Barrilleaux and Grace 2000) . Predicted increases in natural disturbances (shifts in tropical storm and inundation regimes) due to global warming (IPCC 2007a) may favors invasive species by increase resource availability, widen regeneration opportunities, and enhance propogule dispersal (Diez et al. 2012). I anticipate that all of these factors, compounded with increasing anthropogenic movements that may act as a vehicle for short and long distance dispersal (von der Lippe and Kowarik 2007, von der Lippe et al. 2013), will promote encroachment of interior forests by invasive species, with concomitant threats to native plant diversity.

Shifts in Disturbance Regimes and Species Distribution Patterns-

Climate change-induced changes in environmental conditions are likely to affect the distributional ranges of species (Walther et al. 2002), including invasive species (Thuiller et al. 2007), across many regions. With shifts in disturbance regimes (e.g., sea level rise and intensification of tropical storms), coastal ecosystems of southeastern USA will experience altered environmental conditions. Increased salt stress due to salt water intrusion will have greater negative impact on salt intolerant species (e.g., *I. vomitoria* and *M. cerifera*) including invasive *T. sebifera*. After the Hurricane Katrina storm surge flooding, significant mortality of *I. vomitoria* has been reported in leveed bottomland hardwood forests in Louisiana (Howard 2012).

However, a relatively salt tolerant species (e.g., *B. halimifolia*) may persist *in situ* until the sites become inhospitable. I anticipate that the current distributional ranges of salt intolerant native and invasive species will contract farther away from the coast in response to elevated salinity and possibly expand in less saline inland habitats. With respect to inland spread of species, landscape changes and physical disturbances (e.g., canopy damage, inundation, and soil erosion) are likely to determine future community trajectories (Dale et al. 2001). These events increase both environmental (resources, hydrology, and soil types) and biotic (habitat patches, species abundance, and competition) heterogeneity and may promote invasibility. For instance expansion of invasive species in salt stressed mangrove swamps was attributed to disturbance, in particular canopy damage and altered hydrology (Ewel 1986). Similarly, Silliman and Bertness reported that that anthropogenic disturbances and habitat alteration reduce abiotic stresses, increase nutrient loads, and facilitate *Phragmites* invasions in New England salt marshes (Silliman and Bertness 2004). Parallel with these arguments, I expect that tropical storms, compounded with anthropogenic disturbances, will expand regeneration sites, enhance light availability, add nutrients, and increase propagule dispersal of *T. sebifera*, ultimately leading to increases in many previously unoccupied forest lands throughout the southeastern USA. I expect that the expansion and dominance of *T. sebifera* beyond its current range will become an increasingly significant threat to the integrity of coastal ecosystems throughout the southeastern USA as the effects of climate change unfold.

Shifting Storm Regimes and Future of T. sebifera Invasion-

The general view is that hurricane disturbances enhance regeneration of plant species mainly through canopy gap formation (Horvitz et al. 1998, Bellingham et al. 2005, Chapman et al. 2008, Murphy et al. 2008a). However, invasive plant species showed higher regeneration compared to native species in hurricane affected regions (Horvitz et al. 1998, Bellingham et al. 2005, Chapman et al. 2008, Murphy et al. 2008a). Intense hurricanes winds could further increase habitat heterogeneity in forested ecosystems by creating larger canopy gaps which could support the spread of invasive species including *T. sebifera* (Horvitz et al. 1998, Bellingham et al. 2005, Chapman et al. 2008, Murphy et al. 2008a). Extreme events (e.g., hurricanes, flooding, windstorms, and storm surges) may also aid long and short distance dispersal of seeds and thus migration (Diez et al. 2012). It has recently been shown that cyclone Demonia was responsible for dispersing *Parthnium hysterophorus* over long distances, which later threatened agriculture production and wildlife habitat in Swaziland (Burgiel and Muir 2010). Like trails and roads (Sutton et al. 2007), canopy gaps formed along windstorm paths also act as obstruction-free paths for seed dispersal. *T. sebifera* produces large seed crops, which are readily dispersed by birds (Renne et al. 2000, Renne et al. 2001), and I expect a high proportion of *T. sebifera* seeds to be dispersed into canopy gaps and arrive in new sites. Previously, it was reported that a higher percentage of bird-dispersed seeds were dispersed in treefall gaps and from gap to gap but a low percentage in undisturbed forests (Hoppes 1988). Since *T. sebifera* had already begun invading several coastal plant communities, I anticipate further proliferation of this species along the coastal ecosystems as conditions become more conducive with disturbances. When exponential spread of exotic invasive species, after hurricane disturbance, continues to suppress native species growth (Horvitz et al. 1998), we may see novel coastal community assemblages in the near future.

CONCLUSIONS

In conclusion, my results suggest that future increases in soil salinity will likely cause contractions of species' distributions, including *T. sebifera*, at the seaward edges of their current distributions. However, I expect that natural and anthropogenic disturbances in conjunction with landscape structures in more inland areas will create new types of habitats in which native species may not perform as well as invasives (Chapter 3). In a germination study (chapter 3), *T. sebifera* exhibited greater phenotypic plasticity in germination with changing salinity in inland habitats. Plasticity in germination traits may increase the likelihood of seedling survival and establishment (Goode and Allen 2009), and some invasive species are likely to take advantage of disturbed conditions and quickly spread in the landscape. This study further allows us to identify potential sites that are under high risk of *T. sebifera* invasion not only in GBNERR, Mississippi but in coastal forested ecosystems throughout the southeast USA. The increased occurrence of *T. sebifera* juveniles in and around abandoned settlements, active roads, power lines, and water bodies suggests that new invasions in currently un-invaded areas and population explosions of juveniles in recently invaded areas are more likely to occur, leading to future dominance in the forest canopy. Knowing the areas for invasive species probabilities of persistence may help in identifying potential sites where restoration and mitigation processes should be targeted. It can be emphasize that natural disturbances continue to be important events in structuring coastal plant communities experiencing unprecedented exposure to invasive species. Differential occurrence patterns of native and invasive species to environmental factors may generate undesirable shifts in species assemblages, partly by increasing dominance of exotic invasive species and shifting ranges of native species.

Table 2.1. Logistic regression models of the probability occurrence of species in relation to soil water conductivity (COND) and canopy openness (CANOPY).

Species	Parameters of final model	N	Change in deviance[¶]	Terms in simpler supported model (s)
Baccharis halimifolia	Logit *P*= -3.6501+ (0.5583*COND)+ (0.0272*CANOPY) + (-0.0063*(COND*CANOPY))	322	25.0976	COND, CANOPY
Ilex vomitoria	Logit *P*= -0.4187+(-0.0967*COND)+(-0.0006*CANOPY)	398	31.499	COND, CANOPY
Morella cerifera	Logit *P*= -0.8020 +(0.5583*COND) +(0.0105*CANOPY) + (-0.0119*(COND*CANOPY))	448	38.634	COND, CANOPY
Triadica sebifera	Logit *P*= -0.7766+(0.5834*COND)+(-0.0266*CANOPY) + (-0.0105*(COND*CANOPY))	330	5.135	COND, CANOPY

[¶] Change in deviance is reported for comparison between the final supported model versus the simpler model supported by previous steps in the modeling. Reported changes in deviance were greater than the critical χ^2 and significant at the $P = 0.05$.

Table 2.2. Results of logistic regression analysis of some of the environmental factors those are significant in influencing the probability of occurrence of *B. halimifolia*.

Parameters	DF	Wald Chi-Square	P > ChiSq
Sand (%)	1	6.95	0.0084
Silt (%)	1	7.44	0.0064

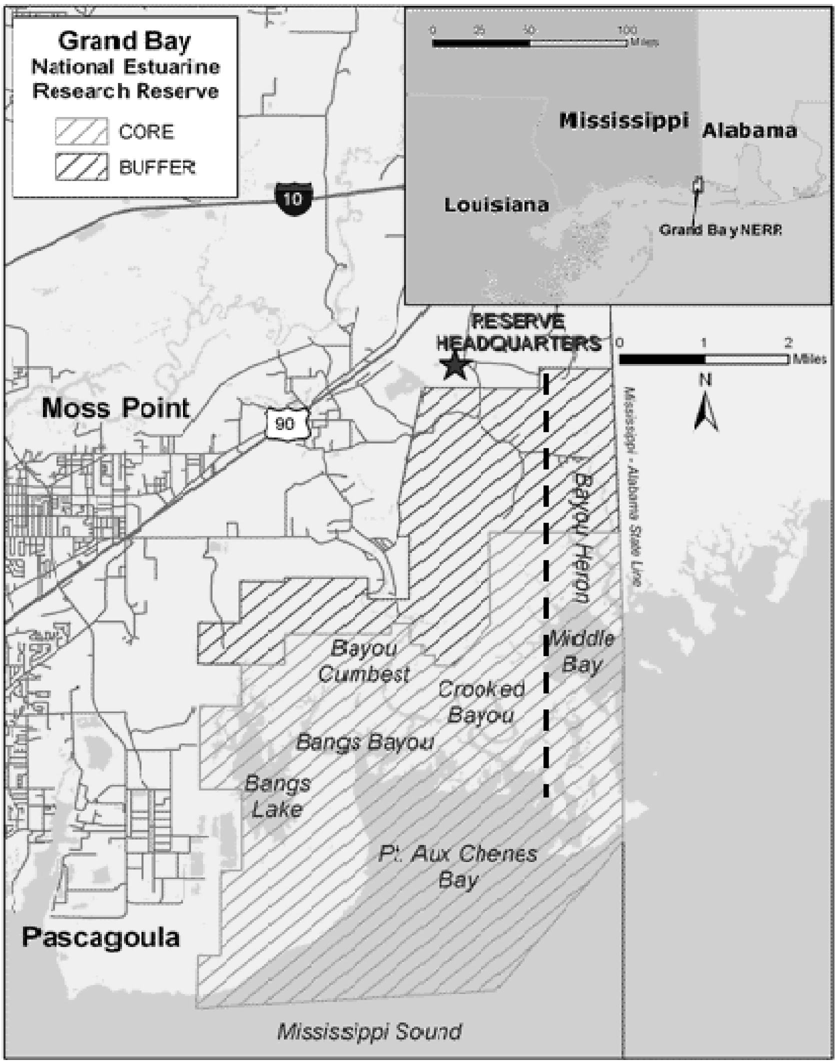

Figure 2.1. A location map of GBNERR (courtesy of the Mississippi Department of Marine Resources) showing 11.3 km long transect (dotted line), which was used as a reference line from which to sample.

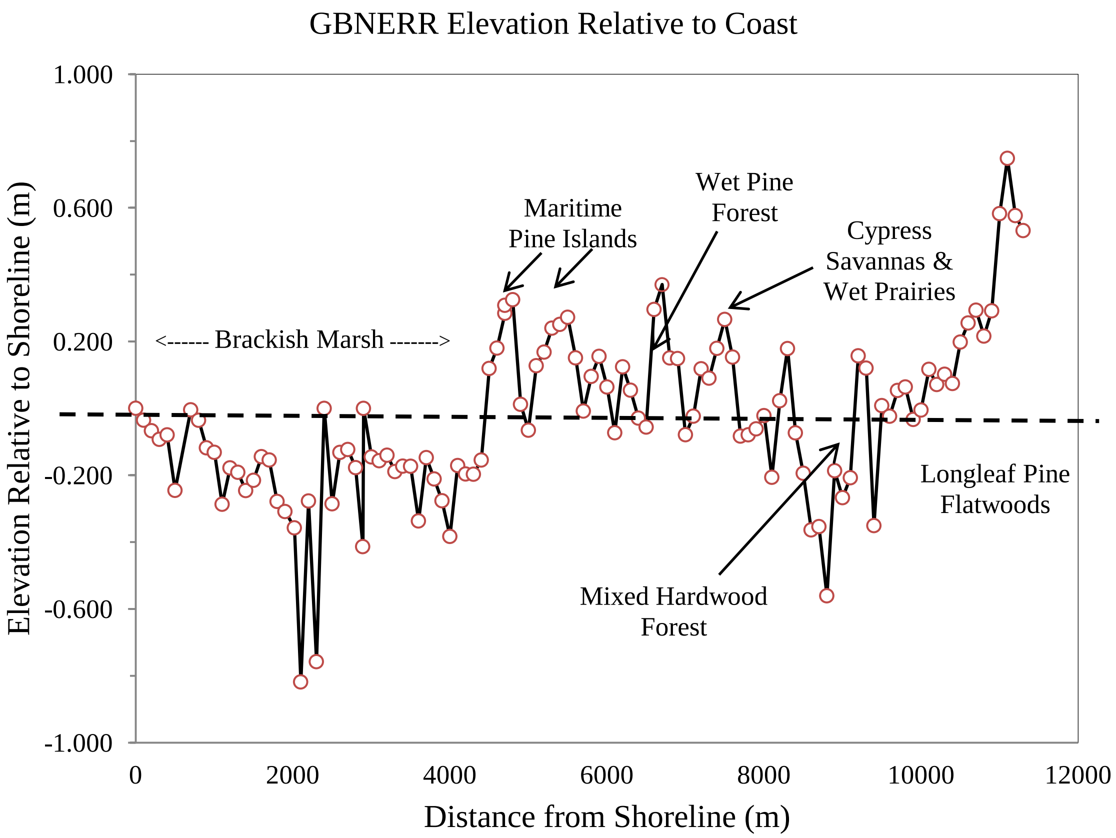

Figure 2.2. Elevation gradient across ~12 Km transect established at GBNERR in 2007 (Battaglia and Platt unpublished).

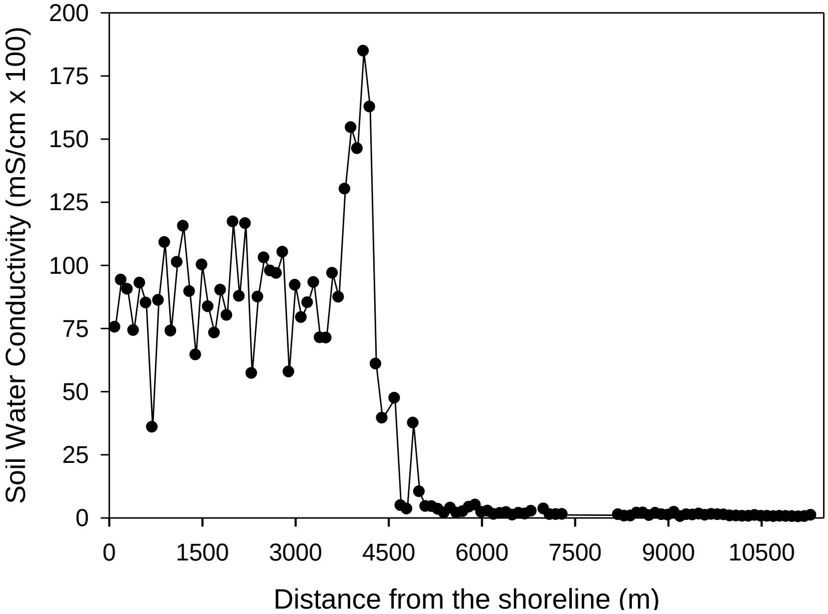

Figure 2.3. Soil water conductivity (salinity) from 2008 survey of 11.3-km research transect that stretches from edge of the sea to inland in GBNERR. Soil water conductivity measurements were a snapshot of one time measurement prior to Hurricanes Gustav and Ike in 2008. Black circles indicate location along the transect from which sampling occurred.

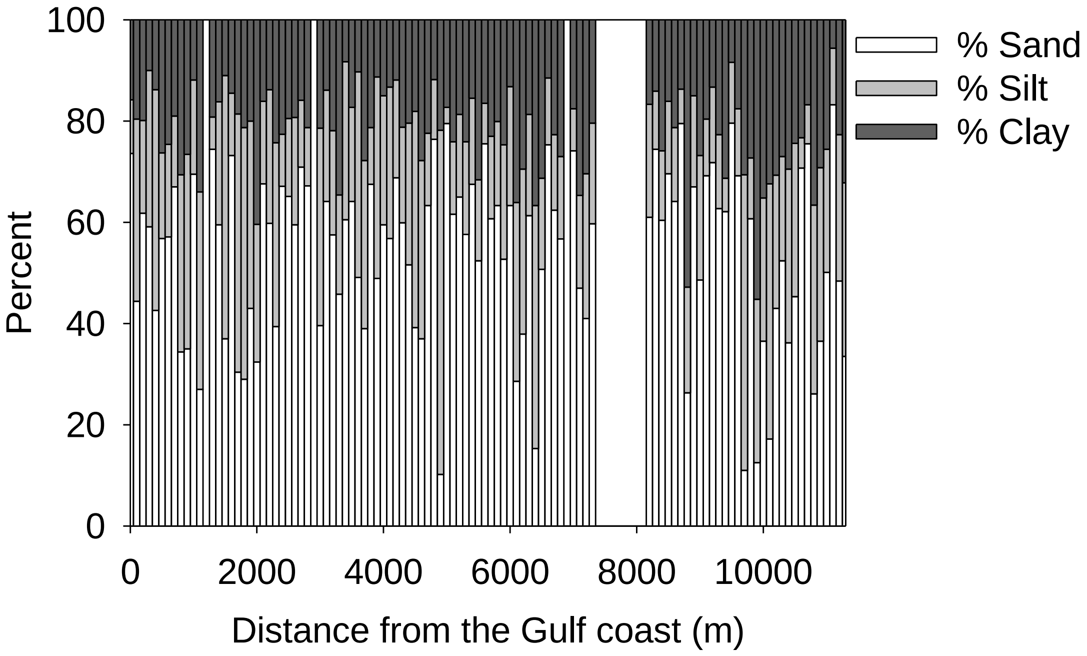

Figure 2.4. Soil texture across the 11.3-km long research transect, which stretches from edge of the sea to inland, from 2008 survey at GBNERR.

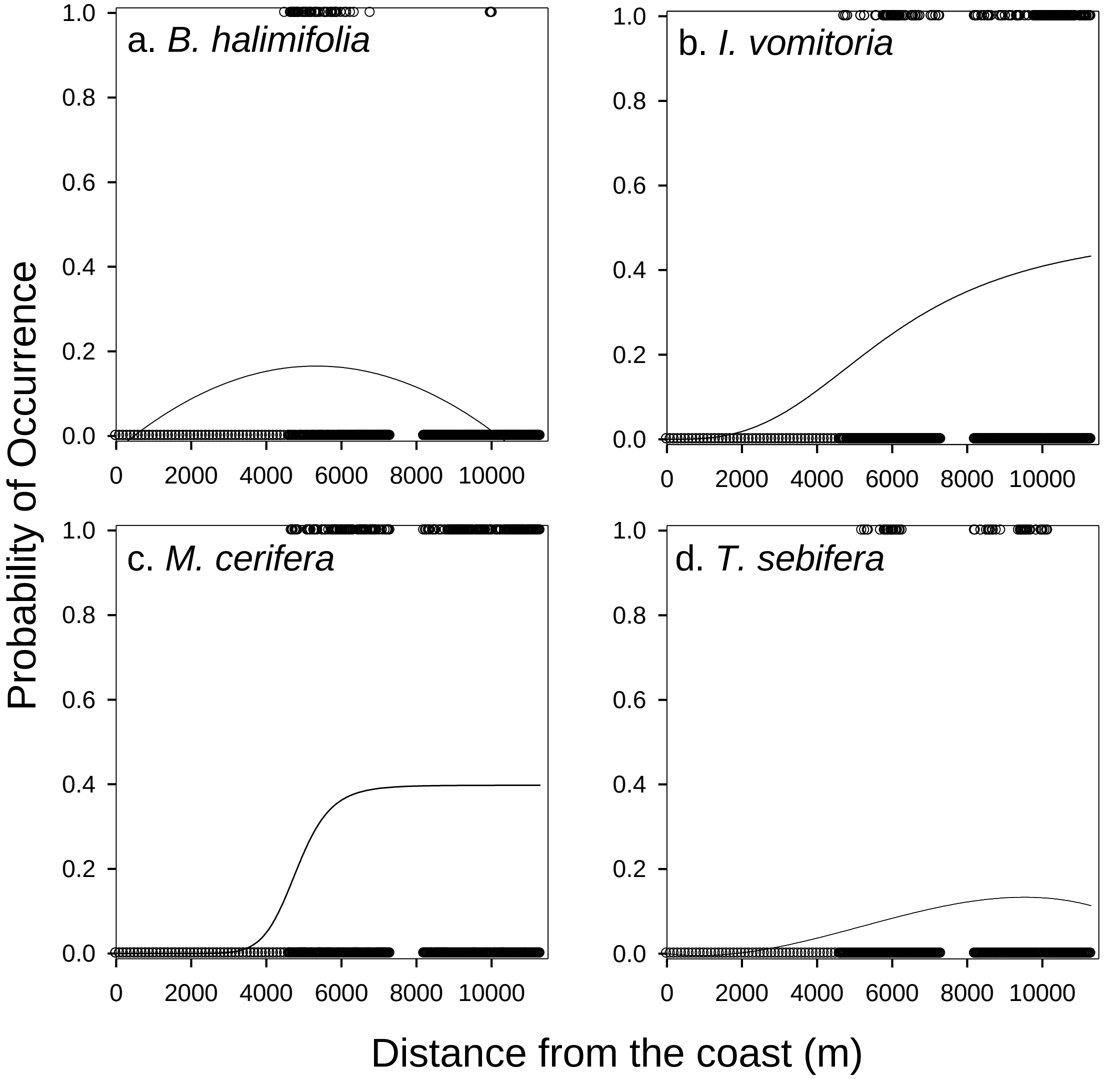

Figure 2.5. Probability of occurrence of species (a) *B. halimifolia*, and (b) *I. vomitoria*, (c) *M. cerifera*, and (d) *T. sebifera* along a 11.3 km long transect at GBNERR.

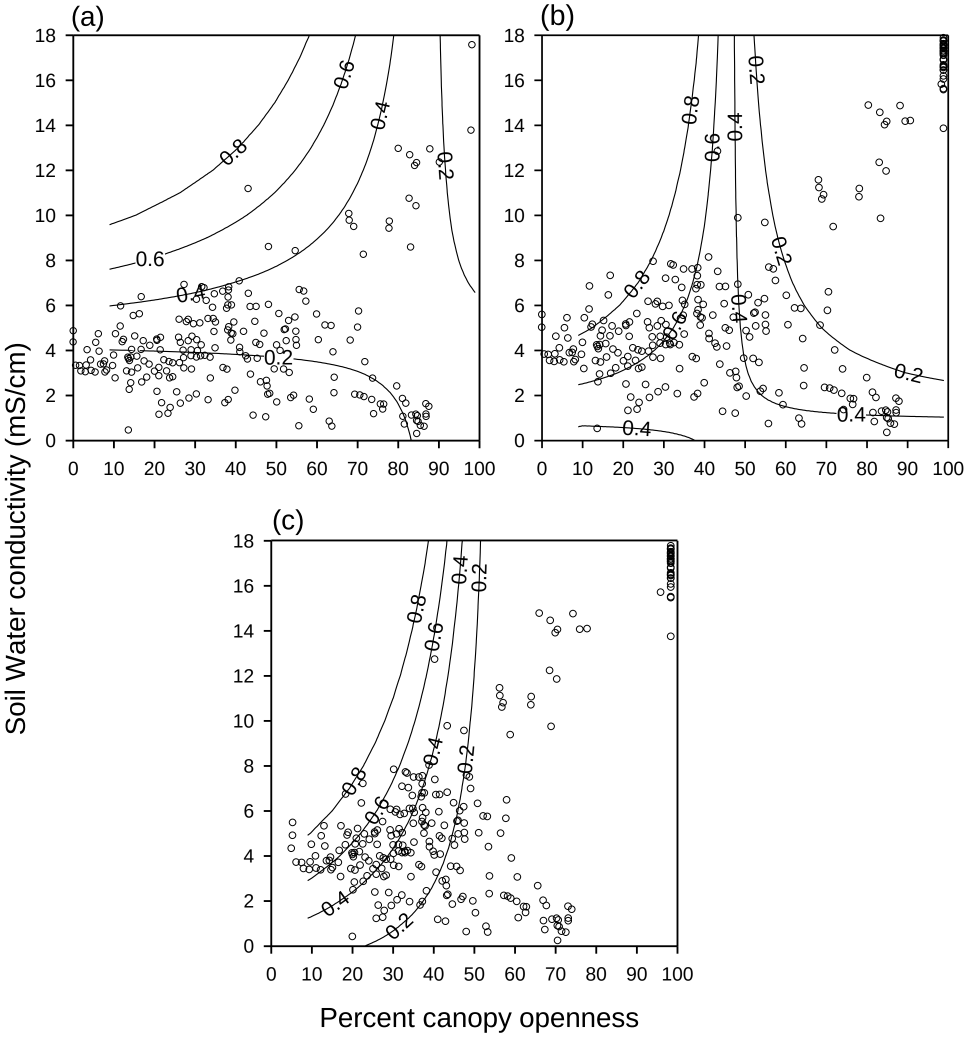

Figure 2.6. Probability of occurrence of species in response to soil water conductivity, percent canopy openness and their interactions: (a) *Baccharis halimifolia*, (b) *Morella cerifera*, and (c) *Triadica sebifera.* (Because species' responses beyond the limits of measured environmental conditions cannot be evaluated, response contours were truncated at the upper and lower limits of the data).

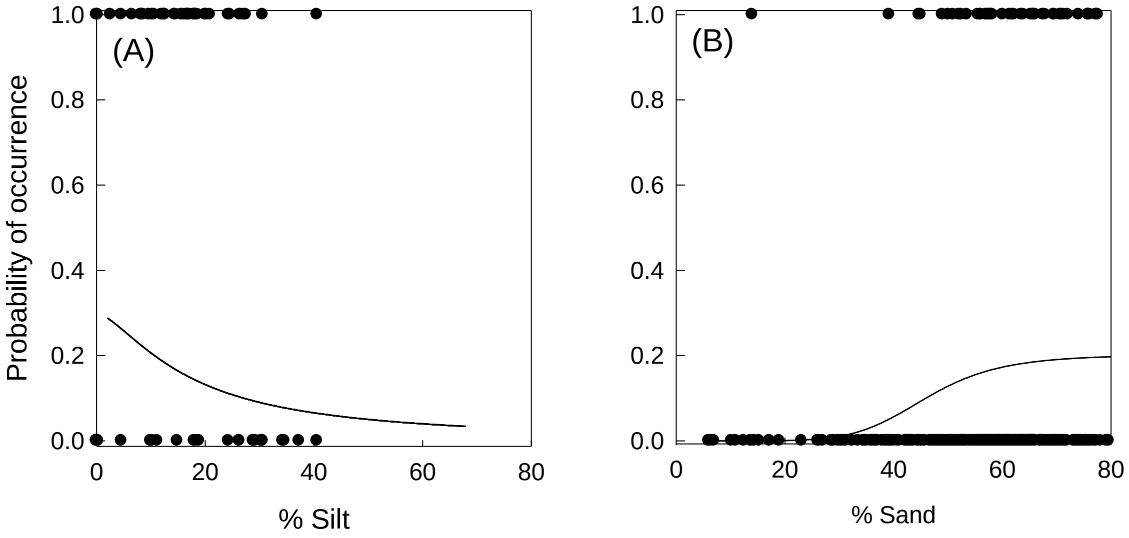

Figure 2.7. Probability of occurrence of *Baccharis halimifolia* in response to (a) percent soil sand and (b) silt.

CHAPTER 3

GERMINATION RESPONSES OF THE INVASIVE TRIADICA SEBIFERA AND TWO CO-OCCURRING NATIVE WOODY SPECIES TO ELEVATED SALINITY ACROSS A GULF COAST TRANSITION ECOSYSTEM

ABSTRACT

With sea level rise, coastal ecosystems farther inland are expected to experience extended duration of inundation. The effects of elevated salinity on initial recruitment of native and invasive species in coastal ecosystems are poorly understood. I assessed effects of elevated salinity across a range of soil types on germination of the invasive *Triadica sebifera* and two natives, *Baccharis halimifolia* and *Morella cerifera*, using growth chamber and greenhouse experiments. In both studies, germination was compared across four salinity levels (0, 10, 20, and 30 g/l). In the greenhouse, percent germination of these species was also compared across soils spanning the five dominant vegetation zones that define a typical coastal transition ecosystem in coastal Mississippi, USA. Germination of all species decreased significantly with elevated salinity in both experiments. In the greenhouse, germination of all species was significantly reduced in soils from the most seaward vegetation zones. Overall, there were species specific responses to salinity treatments; of the three species, *Baccharis halimifolia* was the least sensitive to increasing salinity. With the highest salinity treatment, *Triadica sebifera* germinated well in soils from the most inland vegetation zone compared to other zones, suggesting that saltwater intrusion due to sea level rise may not limit its landward recruitment. Variation in species responses to salinity is likely to drive changes in coastal plant community composition, including distribution shifts and potential landward expansion of *Triadica sebifera*.

INTRODUCTION

Coastal ecosystems, occupying the marine-terrestrial transition, are among the first responders to sea level rise. Rising seas result in a greater duration of inundation and saltwater intrusion in areas farther away from the coast, which in turn alter multiple environmental characteristics including soil moisture, soil and surface water salinity, flooding, and pH of coastal ecosystems (Nicholls and Cazenave 2010). These changes may result in shifts in coastal plant community composition (Michener et al. 1997, Williams et al. 1999a) by increasing the prevalence of stress and/or disturbance tolerant (e.g., invasive and weedy) species (Wang et al. 2006).

Low lying coastal transition ecosystems in the southeastern United States along the northern Gulf of Mexico are highly vulnerable to rising sea levels attributed to the direct impacts of climate change (IPCC 2007b, Titus et al. 2009). Predicted increases in sea level up to1 m or more by the end of this century (Vermeer and Rahmstorf 2009) will result in inundation of many coastal wetlands and increased soil and surface water salinity and magnify the risk of coastal forest retreat (Williams et al. 1999a). Forested wetlands, highly vulnerable to salt stress (Kozlowski 1997), are expected to retreat with chronic sea level rise. Therefore, long term persistence of these coastal plant species depends on their ability to establish at more landward locations.

Tolerance to environmental stresses under natural conditions may provide ecological advantages to species (Wang et al. 2003). Plant species that are more plastic in their tolerance to environmental extremes and disturbances may persist in the landscape, regenerate (Mok et al. 2012), and rise to dominance in the new assemblages. Invasive species often exhibit greater physiological and morphological plasticity compared to native species (Richards et al. 2006), which may enable them to respond more favorably to disturbances and altered environmental conditions (Richards et al. 2006). Further, exotic species, already released from their natural enemies and pathogens (above- and/or below-ground) (Keane and Crawley 2002), may exploit additional regeneration opportunities and devote more resources to growth and reproduction (Daehler 2003) as natives cope with the compounded pressures of environmental stresses due to climate change. For instance, increased plasticity in flood tolerance and positive responses to climatically driven changes in flooding regimes resulted in increased recruitment of *Tamarix ramosissima* in comparison to native *Populus deltoides* in riparian communities in the western US (Birken and Cooper 2006). However, little is known about how the predicted changes in abiotic conditions, associated with chronic saltwater intrusion from sea level rise will affect the recruitment and possible distribution shifts of invasive and co-occurring native species in coastal plant communities. Successful germination is a prerequisite for subsequent growth and development (Donohue et al. 2010), and thus, germination traits are important in determining species’ persistence with climate change (Weber and D'Antonio 1999). Understanding how germination of invasive and co-occurring native species will be affected by increased salinity and inundation across coastal transitions will be necessary to predict trajectories of coastal plant communities in response to sea level rise.

I used germination experiments to investigate how climate change-driven sea level rise and resultant elevated salinity may affect germination of selected invasive (*Triadica sebifera* (L.) Small) and co-occurring native plants (*Baccharis halimifolia* (L.) and *Morella cerifera* (L.) Small) across different coastal vegetation zones. The three species selected for this study are present in coastal ecosystems along the northern Gulf of Mexico, where they are exposed to varying levels of soil salinity and saltwater intrusion. Fine-scale distribution maps of the species indicate that they shared similar habitats along the coenocline at Grand Bay, Mississippi (Chapter 2). Performance comparisons of co-occurring native and invasive species under predicted future climate scenarios should help us to predict the conditions in which the invasive species may proliferate (Daehler 2003).

In this study, the following three hypotheses were addressed. (1) Because elevated salinity negatively affects seed germination of many woody species (Kozlowski 1997), I hypothesized that germination would decline with increased salinity and germination of all species to increase progressively with distance from the coast with declining salt stress. (2) Because salt water intrusion increases soil and surface water salinity, I hypothesized that, irrespective of the locations along the coastal transition, elevated salinity should have negative effects on germination of all the species. Germination of all species is expected to decline progressively with increasing salinity and landward advancement of salinity stress. (3) Because invasive species often exhibit greater phenotypic plasticity than natives (Richards et al. 2006), I hypothesized that *T. sebifera* would have higher germination than native species under all salinity levels.

MATERIALS AND METHODS

Study Site Description-

I selected the coastal transition ecosystem at the Grand Bay National Estuarine Research Reserve (GBNERR) in coastal Mississippi for this study (Figure 3.1**)**. The reserve lies within the gently sloping lower Gulf Coastal Plain, occupies an area of 7284 hectares, and forms one of the most biologically diverse, relatively undisturbed, and productive estuarine ecosystems on the Gulf of Mexico (NOAA 2007). The reserve encompasses many distinct plant communities arrayed along the marine-terrestrial transition. Zonation of plant communities progressively from the coast to inland include: salt marsh, brackish marsh, freshwater marsh, maritime pine forests, coastal pine savanna, mixed pine hardwood forests, pitcher plant bogs, and pine flatwoods.

Plant Species-

Triadica sebifera is a small to medium sized subtropical, deciduous, monoecious tree in the family Euphorbiaceae. It was introduced into the United States in the late 18th century (Randall and Marinelli 1996) and since has become a serious threat to coastal ecosystems in the southeast USA (Bruce et al. 1995, Jubinsky and Anderson 1996). It is spreading rapidly in many coastal communities of the southeast USA (Neyland and Meyer 1997, Wall and Darwin 1999, Denslow and Battaglia 2002, Burns et al. 2004) and causes large-scale ecosystem modification by replacing native coastal prairies and forming monospecific stands of *T. sebifera* in east Texas (Bruce et al. 1995, Jubinsky and Anderson 1996). This species mainly invades abandoned fields, pastures, degraded areas, roadways, recently disturbed sites, and forested wetlands. It reached Mississippi in the mid 1980s and became established in parks, canals, roads, river banks, and similar untended areas (Matlack 2002) and has been spreading rapidly at GBNERR since Hurricane Katrina in 2005. In the reserve, its occurrence is patchy and concentrated in recently disturbed inland habitats (Chapter 2). *T. sebifera* exhibits many traits associated with successful invaders such as fast growth rate, high reproductive rate, reproductive viability in as little as three years (Bruce et al. 1997), large seed mass, effective seed dispersal by birds (Renne et al. 2000, Renne et al. 2001) and water, and a wide range of tolerance to environmental conditions (e.g. flooding and shade) (Conner and Askew 1993, Jubinsky and Anderson 1996, Barrilleaux and Grace 2000), including moderate levels of soil salinity (Conner and Askew 1993).

Baccharis halimifolia is a multi-stemmed deciduous understory small tree or shrub that commonly grows along the Atlantic and Gulf Coast Plains of North America. It grows in moist soils with high organic content such as backswamps, wet prairies, freshwater marshes, and brackish marshes (Penfound and Hathaway 1938, Duncan et al. 1957); and has a moderate level of salinity tolerance (Young et al. 1994). *Morella cerifera* is an evergreen, nitrogen-fixing shrub or tree native to wetlands of the Atlantic and Gulf Coastal plains. This species is shade intolerant and moderately salt tolerant (Duncan and Duncan 1987, Tolliver et al. 1997). At GBNERR, *B. halimifolia* has a patchy distribution and occurs in more saline habitats (such as salt flats); *M. cerifera* has a wider distribution but is restricted to less saline inland habitats (Chapter 2).

Seed Collection and Viability Tests-

Seeds from multiple *T. sebifera*, *B. halimifolia,* and *M. cerifera* individuals were collected in December 2008 from the GBNERR. Collected seeds were mixed and stratified in a cold room (temperature = 4 °C) to mimic natural dormancy periods for two months at Southern Illinois University Carbondale. Before the seeds were placed in a Petri dish and pots for germination, the viability of all species was assessed following guidelines of International Seed Testing Association (ISTA). To remove any fungal infection, which can be present on the surface of the seeds, seeds were surface sterilized by immersing them in a 10% solution of bleach for 30 minutes (*M. cerifera* and *T. sebifera*) and 5 minutes (*B. halimifolia*) and then washed with distilled water. Due to different seed coat characteristics (thick and waxy for *M. cerifera* and *T. sebifera* and thin and soft for *B. halimifolia*), different sterilization time periods was employed. Two samples of 50 seeds of each species were selected randomly and viability tests were performed using 1% 2,3,5-triphenyl tetrazolium chloride (TTC) staining (Porter et al. 1947) as described in (Grabe 1970). Based on TTC staining test results, percent viability of each species was calculated. Results showed that *B. halimifolia* had the highest (65%) seed viability, followed by the invasive *T. sebifera* (42%) and *M. cerifera* (40%). Percent viability of each species was used to adjust the total number of seeds to be used for the germination experiments.

Germination Experiments-

Growth chamber experiment-

To determine the effects of different salinity levels on germination of the selected species, I conducted a controlled growth chamber experiment. In March 2009, based on the viability test results, a fixed number of seeds for each species was placed (50, 77, and 81 for *B. halimifolia*, *M. cerifera*, and *T. sebifera*, respectively) on filter paper (Whatman® qualitative filter paper, Grade 1) at equal distances apart in 6 cm diameter x 1.5 cm deep Petri dishes. The number of seeds was adjusted to ensure the same number (35) of viable seeds of each species in each dish. Seeds in each Petri dish were submerged into 30 ml of saltwater solutions of 0 (de-ionized water - control), 10, 20, 30 g/l (equivalent to 10, 20, 30 ppt, respectively) made using Instant Ocean synthetic sea salt (United Pet Group, Blacksburg, Virginia, USA) mixed with deionized water. These salinity concentrations were chosen to mimic low to full strength seawater field conditions that are expected with sea level rise and resultant saltwater intrusions in low lying coastal transitions of the northern Gulf of Mexico. Petri dishes were wrapped and sealed with parafilm to avoid the loss of water, placed in a growth chamber, and subjected to a regime of 16 hours of light and a day/night temperature of 25/18 °C. These temperature regimes were chosen to mimic local average growing season (late March to late November) temperatures, estimated from monthly long-term data from a weather station at the center of the Reserve (30° 21.551′N, 88° 25.202′W) (National Oceanic and Atmospheric Administration, NOAA, and GBNERR, Mississippi data center). Light (PAR) levels within the growth chamber were 160 µmol $m^{-2}s^{-1}$ and 110 µmol $m^{-2}s^{-1}$for each of the two shelves. Petri dishes were randomly placed in the growth chamber and re-randomized every other day. Petri dishes were inspected every day for the first three months and after that every other week until the end of the experiment (i.e. late October 2009), but no seeds germinated after four months. At each inspection, germinated seeds were counted and removed from the Petri dish. Seed germination was noted when the radicle appeared. During each inspection, water levels were checked for evaporative loss and de-ionized water added to maintain the initial level. The experiment was a complete randomized design with 12 Petri dishes per species (4 salinity treatments × 3 replicates = 12).

Greenhouse experiment-

The greenhouse experiment was conducted using soil collected from the field. Field soil was chosen over commercial soil to represent field conditions with respect to nutrients and other soil properties along coastal transitions. Five dominant coastal vegetation zones at the GBNERR: 1) brackish marsh (BM), 2) maritime pine island (MPI), 3) coastal pine savanna (CPS), 4) mixed pine hardwood (MPH) forest, and 5) pine flatwoods (PFW) (Figure 3.1) were selected. These zones are located sequentially from the edge of the Gulf Coast to the inland in order of BM, MPI, CPS, MPH, and PFW; where BM represents the most seaward zone and is located at 4000 m from the edge of the coast, followed by MPI (5200 m), CPS (6700 m), MPH (9500 m), and the most inland PFW (11200 m). These zones not only differ in distance from the sea and community composition, but also encompass gradients of salinity, flooding, soil properties (such as soil moisture, texture, and nutrients), and elevation relative to the coast. In February 2009, I collected soil samples randomly from the top 10 cm in each of the afore mentioned zones. The samples were brought to the laboratory at Southern Illinois University Carbondale, composited by zone, homogenized by passing the material through a 1 cm sieve, and stored in a cold room (4 °C) until the experiment began in late March 2009.

In the greenhouse, seeds of each species were sown in Deepots (Stuewe and Sons, Inc., Corvallis, Oregon, USA), that were filled with homogenized soil 5 cm from the top, and placed in the supporting rack. Each pot (6.5 cm in diameter × 25 cm tall) had three drainage holes at the bottom. Each supporting rack with nine Deepots (each pot was replicated three times for each species) was placed in a rubber tray (40 cm in diameter × 20 cm tall) filled with the designated concentration of salt water solution. Four trays were assigned to each vegetation zone, each of which was designated to one of four salinities: 0 (no salt addition-control/ambient condition), 10, 20, 30 g/l (equivalent to 0, 10, 20, 30 ppt, respectively) and randomly positioned in the greenhouse. I also drilled a series of 0.48 cm diameter holes into the lower 10 cm and 15 cm of each Deepot to facilitate water flow. Salt solutions were made by dissolving Instant Ocean synthetic sea salt into water (salinity: 3.5 ppt) obtained from Bayou Heron at the GBNERR. A small composite soil-sample was separated from each zone to determine the initial soil salinity. At the end of the experiment, I also collected soil samples from each pot, determined soil salinity, and calculated the final average value for each zone (Table 1). Soil salinity was determined by measuring the electrical conductivity (EC) of a soil solution (electrical conductivity is directly proportional to salinity) with a 1:5 soil-to-water ratio (22 gm: 110 ml)

(Rhoades 1996). The experiment was a complete randomized design with three replicate pots per treatment, with 60 pots for each species (5 zones × 4 salinity treatments × 3 replicates = 60 pots).

As in the growth chamber experiment, a fixed number of seeds, adjusted to achieve the same number of viable seeds (40) for all species, were sown just below the soil surface at equal spacing in individual pots. Water levels were maintained approximately 1 cm below the soil surface in the trays and monitored continuously and controlled by adding deionized water to offset evaporation. Sown seeds were not in direct contact with standing water but received moisture continuously through the soil column. Water salinity was checked every week using a YSI® EcoSense® EC300 conductivity/salinity meter (YSI® Incorporated, Ohio, USA) and maintained for the duration of the study. Pots were inspected for germination every other day for the first two months and every other week until the growing season ended in the field (late November 2009). No seeds germinated after five months. Germination was defined as radicle emerges out from soil. Germinated seeds were counted and removed. Trays and all replicates within each tray were randomized every other week to minimize potential tray and greenhouse position effects.

Statistical Analyses-

Seed germination was expressed as percentage of seeds sown that germinated. For both experiments, germination data were log-transformed to meet assumptions of normality and homogeneity of variance in residuals (Zar 1999). I used a one-way ANOVA for each species separately to test the effect of salinity on germination in the growth chamber (factor = salinity with four levels, completely randomized design). To test the effect of salinity and coastal vegetation zones on germination of invasive and native species in the greenhouse, two- way

ANOVA with interactions (factors = salinity with four levels × five zones) was used. Post-hoc comparisons were performed with Tukey HSD tests when the ANOVA indicated significant differences. All data were analyzed using SAS version 9.2 for Windows (SAS 2008) and results were interpreted at $\alpha = 0.05$.

RESULTS

Growth Chamber Experiment-

B. halimifolia and *M. cerifera* seeds germinated in all salinity levels included in this experiment, but germination decreased significantly as salinity increased (*B. halimifolia*: $F_{3, 11} = 44.42$, $P < 0.0001$ and *M. cerifera*: $F_{3, 11} = 104.93$, $P < 0.0001$). Percentage germination of *B. halimifolia* did not differ between the control (0 g/l), 10 g/l and 20 g/l treatments ($P > 0.05$); and significantly declined at 30 g/l ($P < 0.001$, Figure 3.2). *M. cerifera* germination was significantly reduced in the two highest salinity treatments (Figure 3.2). With the exception of germination at control *vs* 10 g/l ($P = 0.27$), all pairwise comparisons between treatments were significantly different ($P < 0.05$). *T. sebifera*'s germination differed significantly ($F_{3, 11} = 4.63$, $P = 0.037$) among treatments and declined as salinity increased, with no germination at 20 g/l and 30 g/l (Figure 3.2). *T. sebifera* germination was consistently low compared with the other species.

Greenhouse Experiment-

For each species, there was a significant salinity × zone interaction for seed germination (Table 2). Mean percent germination of all species progressively decreased with increasing salinity and the lowest percentage of germination occurred in BM soils (Figure 3.3a,b,c). Mean percent germination of *B. halimifolia* in inland zones (PFW, MPH, and CPS) was significantly higher in ambient conditions compared to all other salinity treatments; the greatest inhibition occurred in 20 g/l and 30 g/l salinity treatments (Figure 3.3a). *M. cerifera* germination varied greatly in ambient conditions across vegetation zones and the greatest inhibition occurred in BM (Figure 3.3b). Germination declined progressively with elevated salinity (10 g/l and above) treatments across the zones (Figure 3.3b). *T. sebifera* germination was significantly lower in BM and did not differ in soils from that zone, regardless of salinity treatment (Figure 3.3c). Under the highest salinity level (30 g/l), this species exhibited significantly higher germination in the most inland zone soils, i.e. PFW compared to other zones, except CPS (Figure 3.3c).

DISCUSSION

Results from both experiments indicate that germination of all species was highest in ambient conditions and decreased progressively with increasing salinity (hypothesis 1). These results agree with previous reports in which seeds of several other wetland plant species were subjected to different salinity levels and exhibited reduced germination (Khan and Ungar 1984, Krauss et al. 1998, Weber and D'Antonio 1999, Gorai and Neffati 2007). Khan and Ungar (1984) found a significant decrease in percent germination of *Atriplex triangularis* with increased salinity stress. Krauss et al. (1998) reported a significant decrease in the germination capacity of *Taxodium distichum* (baldcypress) with elevated salinity in coastal Louisiana.

In this study, the actual pattern of decline differed by species, highlighting the heterogeneity in species responses to elevated salinity. The growth chamber experiment showed that the two native species (*B. halimifolia* and *M. cerifera*) are relatively salt tolerant (up to 20 g/l and 10 g/l, respectively) in terms of germination. In contrast, *T. sebifera* seeds showed higher sensitivity to salinity with a significant decrease in germination at 10 g/l. Results from the growth chamber experiment suggest that chronic saltwater intrusion may be a limiting factor to recruitment of this species in coastal ecosystems experiencing sea level rise (Conner 1994, Barrilleaux and Grace 2000). *B. halimifolia* grows in more saline conditions (e.g., salt flats) compared to other species (Chapter 2), and its salinity tolerance was reflected in the germination study as well (chapter 3). The low germination reported here for *T. sebifera* is somewhat surprising given its seedlings' capability to survive extended saltwater inundation better than some co-occurring native species (Conner and Askew 1993, Howard 2012). A possible explanation is that the long term immersion of seeds in saltwater might reduce the chance of germination by directly disrupting physiological processes (e.g., imbibitions). I therefore anticipate that the elevated salinity due to sea level rise will reduce the potential for regeneration and persistence of the salt intolerant plant species by reducing their germination.

Results from the greenhouse experiment demonstrate that germination of all species was reduced with increasing salinity, as expected (hypothesis 2), indicating that germination will be sensitive to future sea level rise, but the effects will not necessarily be straightforward. Complex interactions between elevated salinity from the advancing sea, local site conditions and species-specific characteristics will drive the responses of native and invasive species. Under ambient conditions, percent germination of species decreased significantly with proximity to the coast, as expected. My results suggest that increased salinity due to sea level rise will further limit germination in zones closer to the coast (e.g., BM and MPI) due to the inherent salinity in their soils (Table 1). These results can be explained with respect to species' current distributions, coupled with the exacerbating effects of elevated salinity, particularly in zones already characterized by saline soils. In the GBNERR, no woody species, including those in this experiments, occur in brackish marsh (chapter 2), probably due to unfavorable, flooded saline conditions. Woody plants are much more sensitive to salinity and flooding during the young seedling stage (Kozlowski 1997), and frequent inundation from tides further reduce suitability of these marshes for establishment of these species. Increased salinity leads to a decrease in the osmotic potential of the soil solution, decreased water absorption by seeds, increased toxicity to the embryo (cf, Kozlowski 1997), and ultimately diminished probability of germination.

Plant species that occur either infrequently and/or are absent from a habitat often exhibit greater sensitivity to climate extremes (Mok et al. 2012). Results indicate that increased soil and surface water salinity due to seawater intrusion will gradually decrease the chance of germination of these species in habitats at the seaward edge of their current distributions. However, germination of species that naturally occur at the most seaward position, such as *B. halimifolia* (Chapter 2), remained relatively high (~ 20%) in10 g/l (Figure 3.3a). This result suggests that *B. halimifolia* is less sensitive at moderate levels of salinity and may therefore persist in its current range along the coastal transition until conditions become too stressful for its recruitment. In a previous study, *B. halimifolia* showed higher salinity tolerance than co-occurring species in coastal estuaries of Delaware (Graves and Gallagher 2003). In contrast, *M. cerifera* is more sensitive and intolerant to salinity (greenhouse experiment), which was also reported previously from a Virginia barrier island (Young et al. 1994), in coastal estuaries of Delaware (Graves and Gallagher 2003), and Santa Rosa Island Florida (Miller et al. 2008). Thus, findings from this study indicate that species highly sensitive to salt stress are likely to be filtered out during the early stage of the life cycle (e.g., germination) and may gradually migrate away from these extremes (i.e., more inland) if dispersal is not limiting. Germination of *M. cerifera* in inland soils under elevated salinity conditions partially supported this prediction, as the species germinated reasonably well (5% or above) with 10 g/l salinity treatments (Figure 3.3b).

T. sebifera maintained approximately 10% germination with salinity of 10 g/l and more than 5% germination with salinity of 20 g/l in inland habitats (Figure 3.3b,c). Interestingly, at the highest salinity level (30 g/l) this species germinated well (mean percentage germination 8%) in the most inland vegetation zone (pine flatwoods) compared to other zones. This study demonstrates that chronic salt water intrusion from sea level rise will not be the lone factor influencing germination of this species. Rather, it is overlain onto inherent differences in soil characteristics that interact with changing salinity to filter germination. Although overall results pointed to *T. sebifera*'s high degree of sensitivity to salinity at the vegetation zones close to the sea, its ability to germinate well at inland locations, even when exposed to increasingly saline conditions, will likely help it to persist and successfully invade (Baker 1974, Honig et al. 1992, Mihulka et al. 2003) coastal forested ecosystems that occupy the more inland portion of coastal transitions in the southeastern US. Invasive species, in general, demonstrate greater phenotypic plasticity that could enhance their tolerance to stress and enable persistence in novel habitats (Walls 2010). It is therefore possible that *T. sebifera* may display plasticity in germination traits in response to altered abiotic environments (e.g., soil and surface water salinity, pH) along the coast, enhancing its regeneration potential. Invading populations of *T. sebifera* have been shown to have greater plasticity to variable moisture conditions and shade compared to native *T. sebifera* populations and its native counterparts (Zou et al. 2009). Greater tolerance of environmental stress that translates into more successful invasion, has been reported for other plant species elsewhere (Daehler 2003). For instance, extended drought tolerance in invasive *Tamarix ramosissima*, compared to co-occurring native species, helps to explain its ability to produce dense stands in Mojave Desert floodplains, western USA (Cleverly et al. 1997). Furthermore, climate-change driven changes in disturbance regimes (e.g., streamflow) are predicted to favor stress tolerant invasive species in riverine ecosystems of western United States (Perry et al. 2012).

Based on germination responses of these species to elevated salinity, I do not expect that *T. sebifera* will replace the other two species, but rather they will coexist. However, extreme disturbance events (e.g., hurricanes), which are predicted to hit coastal ecosystems along the northern Gulf of Mexico (Bender et al. 2010), and anthropogenic disturbances may exacerbate the effects of saltwater intrusion on natural field conditions, particularly if saline storm surges envelop extensive portions of the coastal transition and lead to widespread gaps. Shifts in disturbance regimes will strongly influence the dynamics of coastal plant communities by increasing resource availability and altering abiotic conditions of regeneration sites that may favor invasive plant species (Bellingham et al. 2005, Chapman et al. 2008). For instance, *T. sebifera* seeds are known to germinate well in recently disturbed and moist areas where temperature fluctuations are more frequent (Nijjer et al. 2002, Donahue et al. 2004). Such conditions are likely to be created through canopy disturbances from intense hurricane winds and storm surge (Battaglia et al. 1999, Batista and Platt 2003, Conner et al. 2005), increasing the probability of *T. sebifera* germination and subsequent establishment. Once germinated, *T. sebifera* seedlings are capable of withstanding extended periods of saltwater flooding (Conner and Askew 1993, Jubinsky and Anderson 1996, Barrilleaux and Grace 2000), a trait that may provide it an advantage over less tolerant native species.

Rising sea levels and subsequent increase in soil and surface water salinities are likely to force these species to move upslope/inland along coastal transition ecosystems. However, inland movement and recruitment of these species in response to changing environmental conditions will likely be affected by land-use changes. I expect that successful germination of *T. sebifera* at inland sites is likely to be translated into accelerated establishment, particularly with anthropogenic land use changes that often create invasion foci for exotic species (Dukes and Mooney 1999). Successful germination of *T. sebifera* in various inland habitats and decreased germination of some of the stress-intolerant native species (e.g., *M. cerifera*) are likely to drive rapid compositional shifts in coastal plant communities as the effects of climate change continue to unfold at the marine-terrestrial ecotone.

Table 3.1. Initial (pre-treatment) and final salinity (g/l) of the soils from five coastal vegetation zones at GBNERR, Mississippi. Initial salinity was determined from a composite soil sample representing each zone; while final salinity values are the averages ± SE (n=9 pots) determined at the end of the greenhouse experiment for each zone.

Salinity (g/l) Vegetation zones	Pre-treatment	final at 0 g/l treatment	final at 10 g/l treatment	final at 20 g/l treatment	final at 30 g/l treatment
Brackish marsh (BM)	9.6	9.10 ±0.027	18.80 ±0.380	27.90 ±0.016	38.10 ±0.015
Maritime pine island (MPI)	2.0	1.20 ±0.019	10.10 ±0.021	19.70 ±0.028	29.60 ±0.043
Coastal pine savanna (CPS)	0.20	0.10 ±0.000	10.00 ±0.037	19.00 ±0.039	28.10 ±0.021
Mixed pine hardwood (MPH)	0.0	0.00 ±0.000	9.00 ±0.041	18.80 ±0.053	27.70 ±0.056
Pine flatwoods (PFW)	0.0	0.00 ±0.000	8.60 ±0.029	18.50 ±0.083	26.70 ±0.079

Table 3.2. Results of two-way ANOVA testing the effects of salinity and zones (represent different soil sources from five vegetation zones along coastal transition at GBNERR) on seed germination in the greenhouse.

Source	*df*	*F*	*P*	Source	*Df*	*F*	*P*	Source	*df*	*F*	*P*
Baccharis halimifolia				*Morella cerifera*				*Triadica sebifera*			
Salinity	3, 40	101.39	<0.0001	Salinity	3, 40	61.35	<0.0001	Salinity	3, 40	49.94	<0.0001
Zone	4, 40	17.02	<0.0001	Zone	4, 40	23.92	<0.0001	Zone	4, 40	20.48	<0.0001
Salinity × Zone	12, 40	5.66	<0.0001	Salinity × Zone	12, 40	10.95	<0.0001	Salinity × Zone	12, 40	5.23	<0.0001

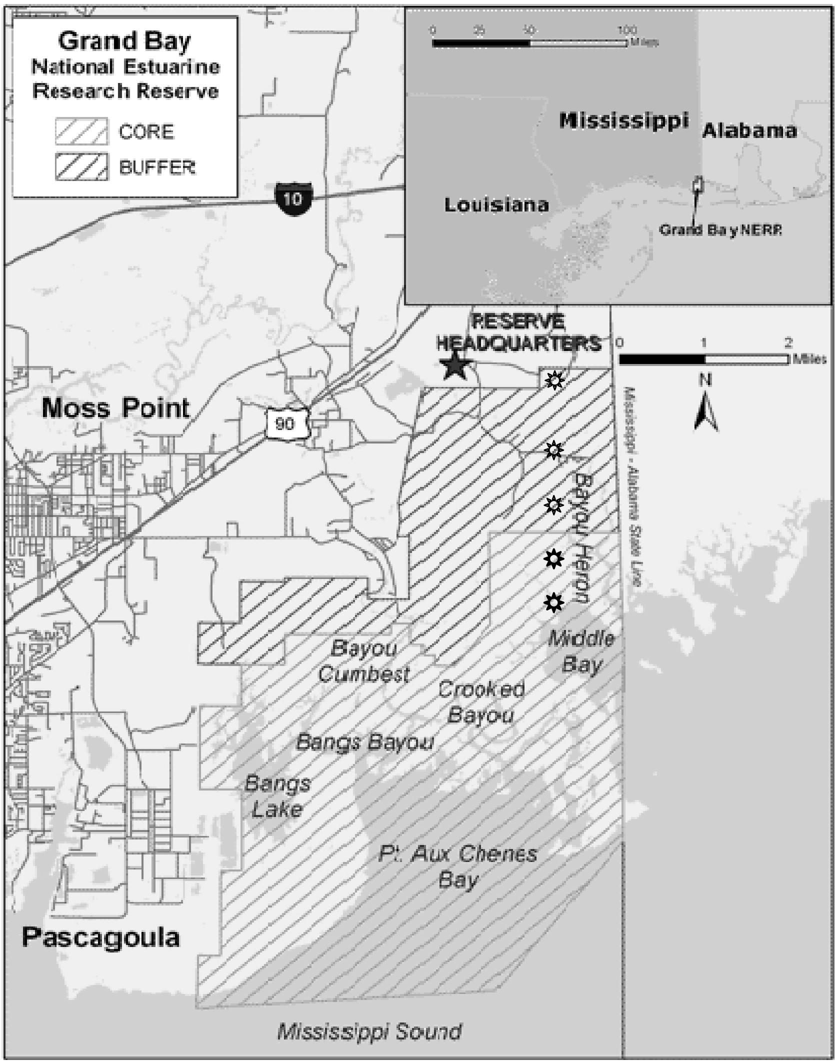

Figure 3.1. A location map of the study area (courtesy of the Mississippi Department of Marine Resources). Symbols (☼) represent five vegetation zones (Brackish marsh (BM), maritime pine island (MPI), coastal pine savanna (CPS), mixed pine hardwood (MPH) forest, and pine flatwoods (PFW), respectively from south to north) along the coastal transition from where soil samples were collected for the greenhouse experiment.

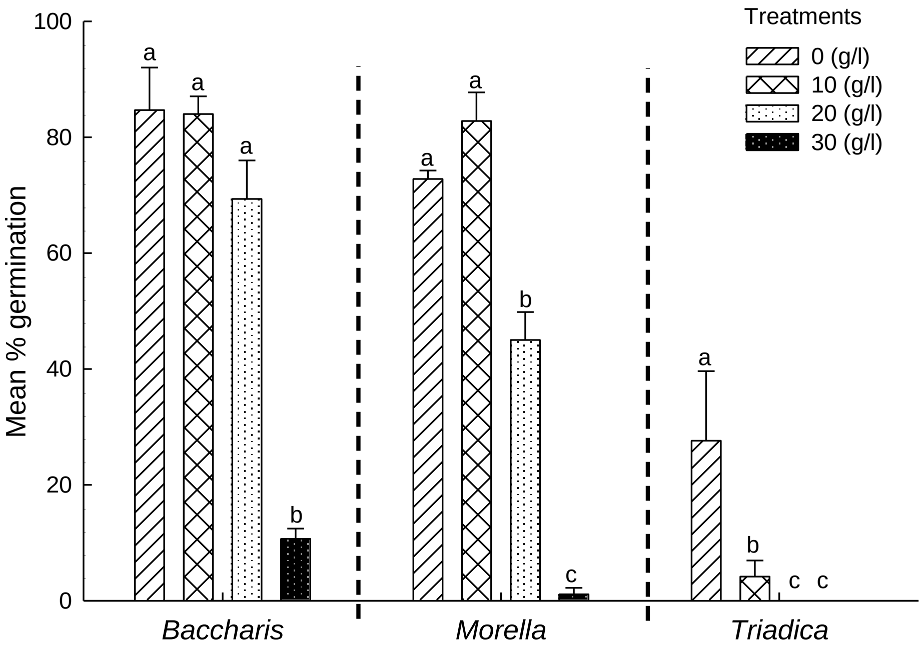

Figure 3.2. Mean percent (± standard error) seed germination at four different salinity treatments in the growth chamber experiment. Bars with different letters represent significant differences among treatments (comparisons of treatment means) in germination within each species, based on Tukey’s multiple means comparisons ($\alpha = 0.05$).

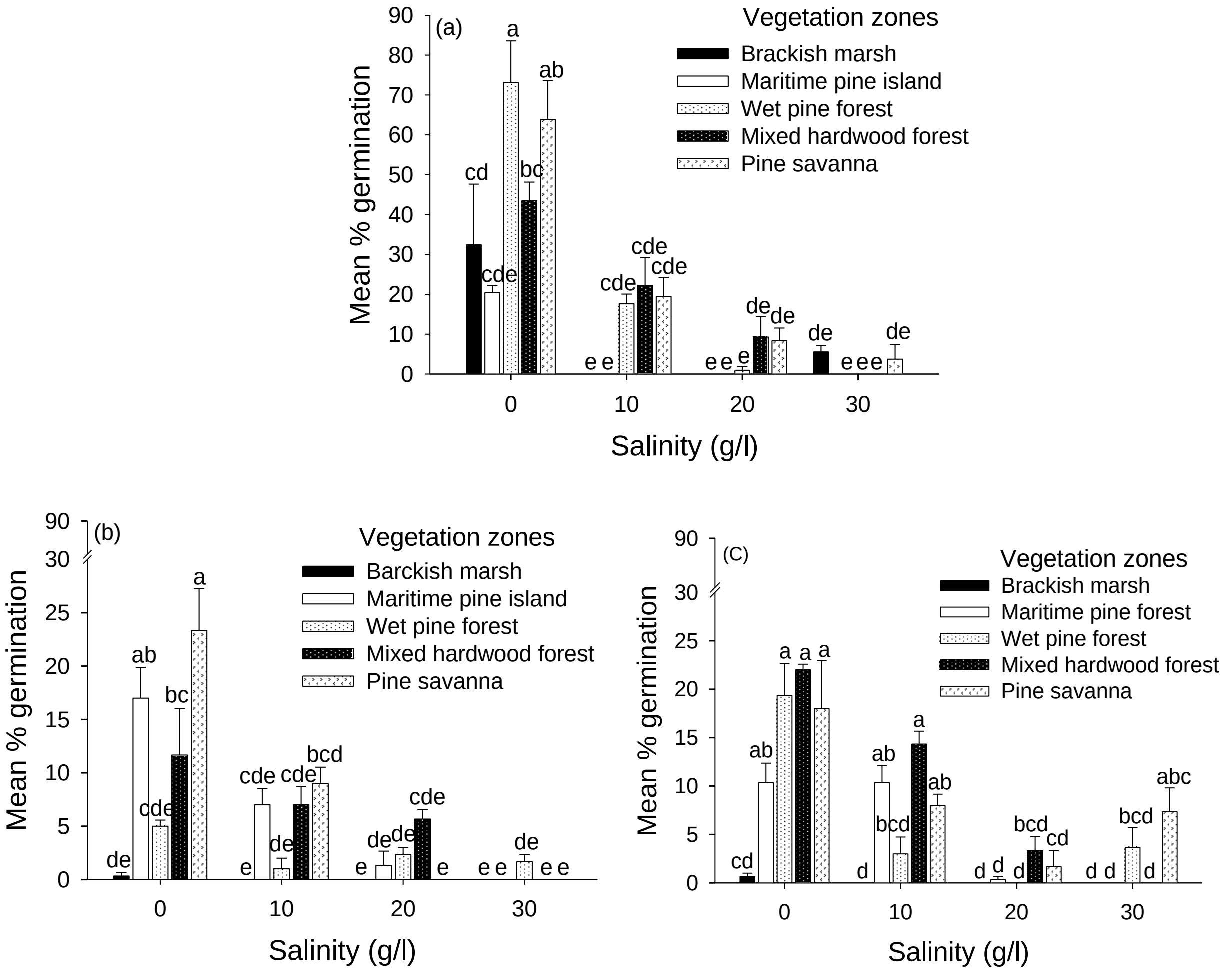

Figure 3.3. Mean percentage germination (± SE) of: a) *Baccharis halimifolia*, b) *Morella cerifera*, and c) *Triadica sebifera* in four different salinity treatments (mimicking control and scenarios of different rates of sea level rise) and five soil types representing five different vegetation zones at the GBNERR in the greenhouse. Bars with different letters are significantly different ($\alpha = 0.05$) (after post-hoc comparisons using Tukey's test).

CHAPTER 4

RESPONSES OF TWO NATIVE SPECIES AND THE INVASIVE *TRIADICA SEBIFERA* TO SIMULATED HURRICANE DISTURBANCES IN FORESTED ECOSYSTEMS OF COASTAL MISSISSIPPI, SOUTHEASTERN UNITED STATES

ABSTRACT

Hurricane disturbances produce significant changes in forest microclimates (e.g., light), creating opportunities for regeneration of plant species and maintenance of diversity. Hurricane winds that damage forest canopies increase light availability, an environmental shift that may provide ideal conditions for rapid growth and spread of invasive species. However, effects of hurricane disturbances are highly variable and species responses to these disturbances are likely influenced by synergistic and idiosyncratic effects of prior site conditions and intensity and direction of hurricane winds. In this manipulative experimental study in the greenhouse, I examined the impact of two key components of hurricane disturbance (canopy damage and storm surge) on two species native to coastal region of the southeastern USA: *Baccharis halimifolia* and *Morella cerifera* and the invasive *Triadica sebifera*. I grew seedlings of these species in soils from the field and under shades, constructed in the greenhouse, which mimicked two forest conditions, Wet Pine Forest (WPF) and Mixed Hardwood Forest (MHF) that are in fire-dominated and fire-suppressed coastal forests of the Gulf Coastal Plain. I manipulated pre-and post-hurricane canopy conditions, applied storm surge, and monitored seedling responses (mortality and growth) to the treatments for sixteen months. Results suggested that species were differentially affected by simulated hurricane disturbances in the two forest conditions and simulated canopy openness increased survival in all the species. Simulated storm surge effects

were short-lived and species recovered their growth under open and low canopy cover. *Baccharis halimifolia* was highly sensitive to shade, whereas *M. cerifera* was sensitive to both shade and storm surge. In contrast, the invasive species *T. sebifera* was resilient to shade treatments in both simulated forest conditions. For all species, simulated storm surge had strong negative effects under high canopy cover particularly in simulated MHF conditions, suggesting storm surge will produce negative impacts on species growth where hurricane winds cause minimal or no canopy damage. Of the three species, *T. sebifera* was by far the most shade tolerant and survival of seedlings under highly shaded conditions may provide it a competitive edge over native species during community reassembly following tropical storms. The patchy nature of these wind disturbances and differential responses of species to disturbances in different forest assemblages across the coastal landscape will eventually drive trajectories of post-hurricane forest recovery and may alter long term structure of coastal forest, potentially favoring the dominance of the shade tolerant invasive *T. sebifera*.

INTRODUCTION

Hurricanes are large-scale natural disturbances affecting structure, composition, and function of coastal forested ecosystems in many parts of the world (Boose et al. 1994, Lugo 2000). Hurricane disturbances produce extensive forest damage through strong winds and inundate coastal and inland ecosystems with salt water through storm surge, producing profound effects on community structure and ecosystem processes (Lodge and McDowell 1991, Michener et al. 1997). Determining the array of plant species responses to the compounded effects of hurricanes (wind storms and storm surges) is an important first step toward understanding how climate change and coincident increasing hurricane intensities may affect coastal forest communities.

Intense hurricane disturbances cause alterations in community structure (e.g., defoliation, branch break, snapped and uprooted trees, accumulation of debris, and soil erosion) and longer-term ecosystem processes (e.g., decomposition, mineralization, etc.) (Lugo 2008). Large scale disturbance events generate heterogeneous environments, widen canopy gaps, reduce competition, and generate growing spaces for early successional fast growing and disturbance tolerant plant species (Pickett and White 1985, Foster et al. 1998, Battaglia et al. 1999). Environmental heterogeneity and increased availability of limiting resources are likely to increase the recruitment of both native and invasive species (Shea and Chesson 2002); increasing prevalence of disturbances and heterogeneous environments often promote recruitment of invasive species (Hobbs 1989, Hobbs and Huenneke 1992).

Effects of hurricanes on coastal plant communities are complex (Michener et al. 1997, Busby et al. 2008, Busby et al. 2009). Plant community responses to these disturbances can also be highly variable and idiosyncratic, often depending on storm and location specific influences such as wind direction, site exposure, local topography, hydrology, soil types, disturbance, community structure (Gardner et al. 1992, Boose et al. 1994, Grove et al. 2000, Kupfer et al. 2008), and site management history (Grove et al. 2000). Studies have demonstrated that stand maturity and hydrology influence susceptibility to hurricane damage (Kupfer et al. 2008, Wang and Xu 2009). For instance, Kupfer et al. (2008) reported that Hurricane Katrina resulted in disproportionately high damage to older forest stands across the southeastern Mississippi landscape. It has also been suggested that disturbed forests with already open canopies have fewer natural windbreaks, as a result, may be disproportionately disturbed by windstorms, increasing environmental and resource heterogeneity in the landscape (Grove et al. 2000).

Hurricane generated storm surge can also result in severe damage to coastal forest vegetation due to salt stress. Increased salt stress from storm surges may have differential ecological impacts, which likely depend on both biotic (i.e., initial species composition and their location with respect to shoreline) and abiotic factors (e.g., topography, the soil conditions), and hurricane intensity (Gardner et al. 1992, Merry et al. 2009). Plants growing on elevated positions far from the coast with highly permeable soils may be less impacted from storm surge than those growing on depressions and less permeable soils (Merry et al. 2009). The storm surge associated with Hurricane Hugo affected forests approximately 1.5 km inland from the forest-marsh boundary in South Carolina (Gardner et al. 1992) and many forests far inland from the Gulf of Mexico coast were inundated and severely stressed following Hurricanes Katrina and Rita, which together produced a storm surge of more than 10 m in some parts of coastal Mississippi (Day et al. 2007, Fritz et al. 2007). However, the effects of salt stress vary greatly by plant species (Conner and Askew 1993, Conner et al. 1997) and recovery of affected species may depend on individual species' salinity tolerance, ability to utilize post-disturbance conditions, and time free from enemies (e.g., pathogens, herbivores). Plant species that exhibit more plastic responses to environmental changes and abiotic stresses, better utilize post-hurricane conditions, and have been released from natural enemies (e.g., exotic invasive) may prevail and grow faster under changed post-hurricane conditions. Studies have shown that, following major hurricanes, exotic plant species regeneration tends to increase dramatically due to increased resource availability, hurricane generated seed dispersal, and decreased resistance of native plant species to disease outbreak and other enemies (Horvitz et al. 1998, Lugo 2008).

The biotic and abiotic effects of hurricane disturbances on regeneration and coastal forest dynamics have been widely recognized and extensively studied (Gresham et al. 1991, Stanturf et al. 2007, Busby et al. 2009, Wang and Xu 2009). Less frequently, investigators have focused on the regeneration patterns of native and non-indigenous species; the few studies suggest that severe hurricanes result in increased dominance of exotic species and suppressed regeneration of native species (Horvitz et al. 1998). However, to my knowledge, no previous studies have examined how co-occurring native and invasive species respond to hurricane associated disturbances (i.e., canopy openings and storm surges) across coastal transition gradients. With increasing tropical storm intensities predicted in the near future (Webster et al. 2005, Bender et al. 2010), there is an immediate need to better understand how species, both native and invasive to low-lying coastal transitions, will respond to increased tropical storm activity. Understanding population- and community-level responses is key to predicting pre- and post-disturbance community structure and vulnerability to invasion.

Low lying coastal wetland ecosystems in the southeastern United States along the northern Gulf of Mexico are highly vulnerable to rising sea levels, tropical storms, and storm surge flooding attributed to the direct impacts of climate change (Battaglia et al. 2012). As a result, these systems are excellent model systems to study how shifts in disturbance regimes will impact native and invasive species performance and response to changing environmental conditions. My approach was to assess the responses of invasive and native species to hurricane disturbances with a focus on Grand Bay National Estuarine Research Reserve (GBNERR) in coastal Mississippi, United States (US), a coastal transition ecosystem typical of the Gulf Coastal Plain. The study system at GBNERR is well suited to address my research objectives because the reserve comprised, among others, contrasting forest stands that differ in several biotic (e.g., community structure/composition) and abiotic characteristics (e.g., distance from the coast, soil texture, and soil salinity) as well as in anthropogenic disturbance histories. I selected two different forest stands: 1) Wet Pine Forest (WPF; hereafter WPF), located approximately 6 km from the shoreline and 1.5 km from the forest-marsh boundary; and 2) Mixed Pine Hardwood Forest (MHF; hereafter MHF), located farther inland (9.5 km from the coast) and 4 km from the forest-marsh boundary. The WPF has comparatively sparse canopy cover (maximum average canopy cover 70%) with an overstory canopy dominated by *Pinus elliottii* and an understory dominated by a mix of *Baccharis halimifolia*, *Ilex vomitoria*, *Ilex*. *glabra*, and *Morella cerifera*, whereas the more inland MHF has a denser canopy cover (maximum average canopy cover 90%) with an overstory dominated by *Pinus elliottii* and other hardwood species and an understory dominated by *B. halimifolia*, *I. vomitoria*, and *M. cerifera*. In recent years, both forest stands have become increasingly infested by the invasive *T. sebifera*, an aggressive invader that has become common throughout the southeastern US. The two most common native species (*B. halimifolia* and *M. cerifera*) and the invasive *T. sebifera* were the foci of this study.

Historically, many ecosystems throughout the southeastern United States Atlantic and Gulf Coastal Plains were impacted by natural fire and hurricane activity (Myers and van Lear 1998). However, in recent years, historical disturbance regimes necessary to naturally maintain these coastal ecosystems have been altered due to anthropogenic fire suppression (Myers and van Lear 1998, Gilliam and Platt 1999). For instance, the MHF stand at GBNERR has long been fire suppressed, resulting in a denser canopy (maximum canopy cover 92%) compared with WPF (Chapter 2). In contrast, the WPF is relatively disturbed with a sparse canopy (maximum canopy cover was 70%) and I personally observed the widespread canopy damage, branch and stem breakage, and tree blowdowns caused by Hurricane Katrina in 2005. Furthermore, WPF has recently been burned while MHF stand has long been fire-suppressed (GBNERR unpublished data). Due to these differential pre-existing environmental conditions and history of disturbances, I expected that responses of native and invasive species following hurricanes would differ across forest stands.

To measure the responses of native *B. halimifolia* and *M. cerifera* and invasive *T. sebifera* to future hurricane disturbances, I established a greenhouse experiment that simulates pre- and post-hurricane abiotic conditions (i.e., light availability, soil salinity) in WPF and MHF stands. The goals of this research were: (1) to elucidate the responses (growth and mortality) of invasive and co-occurring native species to experimental hurricane disturbances in simulated forest stand conditions and (2) to compare the responses of native and invasive species to experimental hurricane disturbances.

MATERIALS AND METHODS

Study System-

The study system is a typical coastal transition ecosystem at the Grand Bay National Estuarine Research Reserve in coastal Mississippi, southeastern United States (Figure 4.1). The reserve lies within the gently sloping lower Gulf Coastal Plain and comprises one of the most biologically diverse and productive estuarine ecosystems in the Gulf of Mexico (NOAA 2007). The reserve encompasses many zoned plant communities arrayed along the marine-terrestrial transition. Coastal plant communities, from the coast inland, include salt marsh, brackish marsh, freshwater marsh, maritime pine forests, wet pine forest (WPF), mixed pine hardwood forests (MHF), and pine flatwoods. This typical coastal transition with distinct forest stands provides an excellent avenue to examine how these ecosystems respond to hurricane disturbances at species levels.

Experimental Setup in the Greenhouse and Data Acquisition-

Seeds of the focal plant species were obtained from GBNERR, representing multiple populations per species, in December 2009. Seeds were mixed and cold stratified at 4°C, a temperature considered suitable for germination in many species with physiological dormancy (Baskin and Baskin 1998); seeds were kept in the cold room at 4°C for three months at Southern Illinois University, Carbondale, IL. After stratification, seeds were surface sterilized by immersion in a 10% bleach solution and then washed with distilled water. Surface sterilized seeds were placed in germinating trays to make a stock of seedlings of all species. In late March 2010, seeds of all species were sown and germinated separately in 52 cm x 26 cm flats, which were half filled with sterile peat moss soil sterilized by autoclaving at 121°C for 45 minutes.

In June 2010, I established greenhouse experiments and applied a canopy openness treatment where I simulated present and expected future environmental conditions (e.g., canopy openness) for two contrasting forest stands, WPF and MHF at GBNERR (Figure 4.1). Twenty pots (26.5 cm diameter by 50 cm height) were allocated for each constructed forest type and were placed on greenhouse benches. Soil samples were collected, separately, from WPF and MHF by digging forty (20 from each stand) holes (30 cm diameter by 50 cm deep). From each hole, I collected soil from the top 10 cm and bottom 40 cm separately. Soil samples were sieved by passing material through a 1 cm sieve and used to fill the pots to within 5 cm of the top. In doing so, I was able to maintain separate field soil columns for the organic (0 - 10 cm in depth) and mineral (10 - 50 cm in depth) horizons in each pot. Half of the pots (10) for the WPF conditions were placed randomly on a greenhouse bench and under a shade house with 30% canopy openness, which mimicked the current lowest average canopy openness recorded at the stand. The remaining pots (10) for simulated WPF conditions were placed on the greenhouse bench with no artificial shade that mimicked 100% canopy openness following hurricane wind disturbance. Full canopy openness (100%) is predicted in the coastal forest stands with sparse canopy following intense hurricanes (Shaffer et al. 2009). To simulate environmental conditions (canopy openness) consistent with MHF conditions, out of 20 pots, half (10) were placed under a shade house with 10% canopy openness, which mimicked the current lowest average canopy openness recorded at the stand. To mimic the future canopy openness following hurricane winds, other pots (10) were placed under a shade house with canopy openness of 70%. Each artificial shade house was constructed with PVC tubes (1.25 m square and 2 m high) and one layer of knitted black shade cloth (FDLAGS) 4.65 m^2 in size and varying in density relative to simulate canopy conditions. A 30 cm gap was left between the bottom of the shade fabric and the bench surface to permit free air movement and reduce temperature differences between the outside and inside of the shade house. One seedling of each species, available from the stock of seedlings, was transplanted into each pot (i.e., three seedlings per pot). Transplanted seedlings were watered as necessary, and pots were arranged randomly in each shade house and reshuffled biweekly to reduce pot effect. Furthermore, during the experimental period (16 months), each shade treatment was moved twice to different greenhouse benches to reduce potential position effects. Approximately halfway through the experiment, the entire experimental setup was moved to a different greenhouse at the same location.

After three months of growth and acclimation periods, seedlings in half of the pots (5) from each canopy openness treatment were surged (storm surge treatment) with full strength sea water (30 ppt) to mimic a storm surge event. This surge was designed to simulate a typical hurricane surge occurs in coastal areas close to shoreline (Macauley et al. 2007). Storm surge was accomplished by applying an appropriate concentration of salt water solution (prepared by dissolving synthetic Instant Ocean® (Spectrum Brands, Madison, Wisconsin, USA) sea salt into the water) into the pots. The surge was maintained for three days at 10 cm above the soil substrate in the pots. The remaining five pots in each canopy treatment were not surged (i.e., control). After three days, water from the surged pots was siphoned out and pots were flushed with fresh water. Height of growing seedlings in each pot was measured at the beginning of the experiment and bimonthly before simulated storm surge events. After the surge, seedling heights were measured bimonthly. Mortality of all species within each pot was also recorded. After a 16 month period that covered almost two full growing seasons, all surviving plants were harvested. Each plant was separated into belowground (roots) and aboveground (shoot) parts; roots were rinsed with distilled water to remove soil particles. Shoots and roots were dried to a constant temperature at 55°C, and then weighed. Dry weight was used to determine shoot and root biomass.

Statistical Analyses-

A contingency table analysis was performed to test whether there is an association between simulated canopy openness and storm surge treatments on mortality for each of the studied species in each forest conditions. Since all the expected frequencies were less than or equal to 5, I used Fisher's exact test, which does not display test statistics but gives a direct *P*-value (Upton 1992). Changes in growth were expressed as relative growth rates (RGRs) and was calculated as $RGR = (\ln w_2 - \ln w_1)/(t_2 - t_1)$, where w_1 and w_2 is the plant height at time t_1 and t_2, respectively. I used a repeated measures mixed model for each species separately to test the effect of simulated storm surge, time, and their interactions on RGR (factors = storm surge two levels × month) within each simulated canopy openness. Post-hoc comparisons were performed with Fisher's LSD multiple comparisons among individual treatments when the results indicated that there were significant overall differences. Total dry biomass (shoot + root) of all species were compared between simulated surged and un-surged treatments within the simulated canopy openness using one way ANOVA. Total dry biomass data for all the species from simulated canopy openness were transformed using the log function to meet assumptions of normality and homogeneity of variance in residuals (Zar 1999). All data were analyzed using SAS version 9.2 (SAS 2008) and results were interpreted at $\alpha = 0.05$.

RESULTS

Seedling Survival and Growth Responses-

Wet Pine Forest (WPF) Conditions-

Seedling mortality data suggests that, under simulated 30% canopy openness, *B. halimifolia* experienced a higher proportion of seedling mortality in unsurged pots relative to surged pots (Table 4.1). Under simulated 100% canopy openness, however, seedling mortality occurred only in some of the simulated storm surge pots (Table 4.1). In 30% canopy openness, *M. cerifera* seedlings experienced mortality in both simulated storm surged and unsurged pots (Table 4.1). However in simulated 100% canopy openness, seedling mortality occurred only in simulated storm surged pots. For *T. sebifera* the proportion of seedling mortality was 0.8 in storm surged pots and 0.4 in unsurged pots under simulated 30% canopy openness. Similar to other species, seedling mortality was recorded only in some simulated storm surged pots in 100% canopy openness (Table 4.1).

For *Baccharis halimifolia*, there was a significant storm surge × time interaction effect on the relative growth rate (RGR) (Table 4.2; $P < 0.0001$) under simulated 30% canopy openness. Overall, seedlings growth declined gradually during the first growing season; thereafter it did not change significantly (Figure 4.2A). Initially, RGR of the seedlings that received storm surge treatment differed significantly from unsurged ones (until Oct-10) but the differences were not significant thereafter (Figure 4.2A). In 100% simulated canopy openness, simulated storm surge did not have a significant effect on RGR (Table 4.2; $P > 0.05$). The RGR declined gradually during the first growing season; thereafter it did not change significantly (Figure 4.2B). For *Morella cerifera* and *Triadica sebifera* in 30% canopy openness, RGR declined gradually during the first growing season and did not change significantly thereafter, (Figure 4.3A and 4.4A, respectively) and simulated storm surge had no effect on RGR (Table 4.2). There was a significant storm surge × time interaction effect on the RGR in simulated 100% canopy openness for both species (Table 4.2; $P < 0.01$). Initially, seedlings that received storm surge had significantly lower RGR compared to unsurged ones, but the differences were not significant thereafter (Figure 4.3B and 4.4B, respectively for *M. cerifera* and *T. sebifera*).

Mixed Hardwood Forest (MHF) Conditions-

Under simulated 10% canopy openness, regardless of simulated storm surge, *B. halimifolia* and *M. cerifera* experienced 100% seedling mortality (Table 4.1). Under simulated 70% canopy openness, some seedlings of both species experienced mortality in simulated storm surged pots (Table 4.1). In contrast, some of the *T. sebifera* seedlings under simulated 10%

canopy openness in unsurged pots survived and remained until the conclusion of the experiment. Similar to other species, some seedlings of *T. sebifera* experienced mortality in simulated storm surged pots under simulated 70% canopy openness (Table 4.1).

Since simulated canopy openness (10%) resulted in mortality of all seedlings (regardless of storm surge) for both *B. halimifolia* and *M. cerifera* within a few months of the experiment, the measurement of growth responses over the course of the experiment was not possible for that canopy treatment. In simulated 70% canopy openness, for both species, storm surge had no significant effect on RGR (Table 4.2; $P > 0.05$). RGR of both species decreased progressively during first growing season, but the growth rates were increased significantly during the early part of the second growing season (Figure 4.5A and Figure 4.5B, respectively for *B. halimifolia* and *M. cerifera*). All *T. sebifera* seedlings grown in simulated 10% canopy openness and subjected to simulated storm surge also died after a few months. Results showed that the RGR of *T. sebifera* seedlings in the unsurged pots decreased progressively during the first growing season, but the growth rate did not differ thereafter (Figure 4.6A)). However, there was a significant storm surge × time interaction effect on RGR in simulated 70% canopy openness (Table 4.2; $P = 0.015$). Initially, seedlings that received storm surge had significantly lower RGR compared to unsurged ones, but the seedling recovered well during second growing season (Figure 4.6B).

Seedling Biomass Responses to the Treatments-

Wet Pine Forest (WPF) Conditions

Mean total biomass of *B. halimifolia* under simulated 30% canopy openness was less (4.73 gm; SE = 3.59) than under 100% openness (52. 53 gm; SE = 14. 10) (Figure 4.7A). Similarly, for *M. cerifera* mean total biomass in simulated 30% canopy openness was 0.58 gm (SE = 0.25) and in simulated 100% canopy openness was 13.32 gm (SE = 3.38) (Figure 4.7B). Mean total biomass of *T. sebifera* under 30% canopy openness was 10.48 gm (SE = 4.10) versus 13.52 gm (SE = 3.98) under 100% canopy openness (Figure 4.7B). In both simulated 30% and 100% canopy openness, I did not find significant effects of simulated storm surge on total biomass of all the species (*P* > 0.05; Table 4.3).

Mixed Hardwood Forest (MHF) Conditions-

Regardless of simulated storm surge treatment, simulated 10% canopy openness resulted in mortality of all *B. halimifolia* and *M. cerifera* seedlings within the first half of the study period, so a biomass harvest was not possible for those species and potential effects of storm surge under simulated 10% canopy openness could not be tested (Table 4.3). In simulated 70% canopy openness, simulated storm surge did not have significant effects on total biomass of *B. halimifolia* and *M. cerifera* (Table 4.3; *P* > 0.05). In contrast, some of the *T. sebifera* seedlings survived in unsurged pots within the 10% canopy openness. However, potential effects of storm surge could not be tested due to the 100 percent mortality of seedling from storm surged pots. In 70% canopy openness treatment, I did not find a significant effect of simulated storm surge on total biomass (*P* > 0.05; Table 4.3).

Mean total biomass of *B. halimifolia* under simulated 70% canopy openness was 10.86 gm (SE = 2.46). Mean total biomass of *M. cerifera* under 70% openness was 3.87 gm (SE =

1.17). For *T. sebifera* mean total biomass was less (1.56 gm; SE = 0.13) in simulated 10% canopy openness than in simulated 70% canopy openness (11.25 gm; SE = 2.36) (Figure 4.8).

DISCUSSION

The responses of two native species and the invasive *T. sebifera* to key components of simulated hurricane disturbances (canopy damage and storm surge) demonstrates that both factors are potentially important determinants of plant performance. Simulated hurricane disturbances produced differential effects in simulated forest conditions and the species growing under those conditions demonstrated heterogeneity in their responses. In both simulated forest conditions and under simulated increased canopy openness (greater light availability), I recorded a higher proportion of seedlings survival and more total biomass. In particular, I found marked differences among species in seedling survival and growth following simulated storm surges in simulated MHF conditions. Simulated storm surge generally had detrimental effects on seedling survival and growth, but effects were much more pronounced in MHF conditions. In MHF conditions, the combination of storm surge and heavy shade resulted in mortality of all species, suggesting that storm surges with little canopy damage produce substantial negative impacts on seedling regeneration of woody species in hurricane impacted coastal areas. Thus, it is likely that it was the improved light availability following canopy opening due to hurricane winds that increased performance of young seedlings (e.g., survival and growth) of both native and invasive species (Bellingham et al. 2005, Chapman et al. 2008, Murphy et al. 2008b, Shiels et al. 2010). However, performance of these species under natural field conditions would likely be determined by interactive effects with other key components of hurricanes, such as storm surge (Middleton 2009, Howard 2012) and litter and detritus deposition (Murphy et al. 2008b, Shiels et al. 2010). Furthermore, winds and storm surge are not necessarily linked; exceptionally strong storms result in sustained storm surges throughout the Gulf of Mexico coast (up to 100 km from where they make landfall; Fritz et al. 2007) but canopy effects are more local to the storms' direct paths.

Seedling Responses to Simulated Canopy Openness Treatment-

Canopy disturbance increases light availability into the forest understory, which often results in enhanced regeneration and growth of plant species (Canham and Marks 1985). Although the patterns of responses differed between simulated forest conditions and among species, increased canopy openness increased seedling survival and increased total biomass for all species at both forest conditions. These results are comparable with several previous studies that demonstrated the positive relationships between increased canopy openness (both from natural and simulated hurricane disturbance) and seedling survival, growth, and abundance in many hurricane impacted regions (Battaglia et al. 1999, Busby et al. 2009, Merry et al. 2009, Shiels et al. 2010). Shiels et al. (2010), following experimentally simulated hurricane disturbance (e.g., canopy openings) in the field, recorded a significant increase in growth and recruitment of woody seedlings in subtropical wet forests in Puerto Rico. Increased sapling growth following simulated hurricane disturbances was also reported in New England forests (Cooper-Ellis et al. 1999). It has been well documented that canopy openings from hurricane winds have substantially favored the dominance of fast growing non-native species in various hurricane impacted regions (Bellingham et al. 2005, Snitzer et al. 2005, Murphy et al. 2008b). Murphy et al. (2008) documented a significant increase in seedling abundance of invasive plant species following Cyclone Larry in Australian rainforests. Following Hurricane Andrew, reduced regeneration of native vine species but rapid expansion of invasive vines was reported from Florida (Horvitz et al. 1998). In contrast, this experiment produced mixed results on the performance of invasive *T. sebifera*, as its performance (e.g. biomass) did not change much under simulated WPF conditions after simulated canopy opening. In contrast, its survival and biomass was higher substantially after simulated canopy openings consistent with conditions at MHF. It is anticipated that post-hurricane regeneration of *T. sebifera* likely to be vary with respect to pre-disturbance forest conditions in along the coastal transitions.

In this study, both native species *B. halimifolia* and *M. cerifera* benefitted from simulated increased canopy openings, exhibiting higher proportion of seedling survival in both simulated forest conditions and have higher mean total biomass. This finding is consistent with recent studies that report increased growth of native *Acer saccharum*, along with invasive *Acer platanoides* seedlings following elevated light intensity in forests of the Lawrence River Valley of Montreal, Canada (Lapointe and Brisson 2012). Natural and anthropogenic disturbances may increase resource availability which increase the performance of plant species (Hobbs and Huenneke 1992). My results suggest that increased light, as a direct result of hurricane induced canopy damage, can substantially increase the performance (e.g., survival and biomass) of native species in forestlands of coastal Mississippi and that responses would vary with respect to pre-disturbance forest conditions. However, under field conditions, success of a species in post-disturbed conditions will also depend on differential species' efficiencies in uptake of unused and/or newly available resources (Davis et al. 2000), differential species' competitive advantage due to release from their natural enemies (Mitchell and Power 2003, DeWalt et al. 2004), and the availability of propagules for regeneration.

Triadica sebifera has many characteristics typical of noxious invasive species, including rapid growth rates (Bruce et al. 1997), early reproduction (in some cases, flowering within a year of germination; JB Grace, *pers. comm.*), copious propagules production and efficient propagule dispersal (Renne et al. 2002), and rapid adaptation to insect herbivores in its introduced range (Siemann et al. 2006). *Triadica sebifera* is also tolerant of a wide range of environmental conditions, including closed canopy understories (Jones and McLeod 1989), storm surge flooding (Howard 2012), and moderate levels of soil salinity (Conner et al. 1997). The combination of favorable vital attributes with a broad fundamental niche likely provides a competitive edge to this species under altered environmental conditions. In this study, some *T. sebifera* seedlings survived in simulated current MHF canopy conditions (10% canopy openness), while native seedlings had 100% mortality. These results suggest that invasive *T. sebifera* will have more opportunities to exploit post-hurricane environments while native species may not have seedlings in place when gaps become available. Substantial increases in invasive species' density and coverage following hurricane disturbances have been documented elsewhere (Horvitz et al. 1998, Murphy et al. 2008b), including cases involving *T. sebifera* in coastal Louisiana (Chapman et al. 2008, Howard 2012). Once *T. sebifera* survives high shade conditions, it has the ability to grow rapidly under moderate to full sunlight following canopy openings from hurricane winds (Jones and McLeod 1989). Phenotypic plasticity and stress tolerance is one of the major traits that make invasive species successful in invaded habitat (Rejmanek and Richardson 1996, Richards et al. 2006). Here, I expect that high shade tolerance of *T. sebifera* can provide this species a competitive advantage over native species under post-hurricane environments and I anticipate that this aggressive invader will increase its current range along the forested wetlands of coastal southeastern US with increased intensity and severity of hurricane disturbances predicted for this region (Bender et al. 2010).

Seedling responses to simulated storm surge-

The responses of the species to storm surge were not uniform between the simulated forest conditions. *Baccharis halimifolia* seedlings experienced low mortality, *M. cerifera* and *T. sebifera* seedlings experienced more mortality from simulated storm surge in simulated WPF conditions. However, for all species the total dry biomass was not different between simulated storm surge and unsurged treatments. In contrast, storm surge produced 100% mortality of all species in 10% canopy openness in MHF conditions. Under natural conditions, the effects of storm surges are expected to be more severe in young seedlings due to salinity stress, scouring, and erosion that exposes root systems, leading to desiccation stress (Stanturf et al. 2007). In this study, I simulated only the elevated salinity (30 ppt) for three days and some ecological effects were ephemeral, particularly in less shaded conditions, where seedlings of all three species recovered their normal growth rates. Thus, I predict that seedling mortality in 30% and10% canopy openness under WPF and MHF conditions, respectively may be associated more with shade than the surge effects. Still, in areas with dense canopy cover, storm surge effects are likely to be exacerbated due to the dual and interactive stresses of shade and salinity.

Results from this study suggest that forest stands that remain relatively undisturbed by hurricane winds may be more susceptible to hurricane generated storm surges. Thus, the patchy nature of hurricane wind disturbances to forest canopy (Stanturf et al. 2007, Kupfer et al. 2008) likely produces heterogeneous effects of storm surge by differentially affecting the survival and growth of seedlings in different habitats along the coastal forests. In coastal Mississippi, substantial land cover changes (expansion of open water and reduction in vegetated areas) have already been recorded as a result of the unprecedented effects of storm surges following Hurricane Katrina and other recent storms (Evans et al. 2012). Furthermore, widespread mortality of understory and overstory trees in coastal bottomland hardwoods in Louisiana after extended duration of storm surge flooding (up to 3 weeks) from Hurricane Katrina, which produced minimal canopy damage, have recently been reported (Howard 2012). Thus, storm surge from hurricanes is likely to have differential effects on survival and regeneration of native and invasive species in low-lying coastal forestlands, leading to compositional shifts in coastal plant communities.

Complex Interactions and Outcomes of Differential Species Responses-

In this study, I found that native and invasive species' responded differently to simulated hurricane disturbances at different forest conditions and across time. Overall, all three species demonstrated positive responses to simulated canopy openings through a higher proportion of seedling survival and an increased biomass. Results showed that the negative effects of storm surge are more intense under high canopy cover in both WPF and MHF conditions, suggesting that the canopy structure and storm surges may act synergistically, and the interactions between these factors may ultimately direct species' responses. In low-lying coastal areas with sparser canopies and relatively saltier soil (i.e., WPF), *B. halimifolia* may benefit more than other species because of its greater tolerance to salinity (Chapter 2). It is likely that highly shade tolerant *T. sebifera* will increase in dominance in MHF and other areas where historical natural disturbance regimes have been altered through anthropogenic intervention. For centuries, many of the ecosystems throughout the Coastal Plain of the southeastern United States have been maintained by natural fire and hurricane activity (Myers and van Lear 1998). During the post settlement era, however, anthropogenic alteration of the historical disturbance regimes necessary to maintain these coastal ecosystems has resulted in structural and compositional shifts in plant communities (Myers and van Lear 1998, Gilliam and Platt 1999). For instance, field observation and canopy photo analysis (Chapter 2) indicated that the forest stand at MHF has long been fire-suppressed, resulting in a denser canopy (canopy cover reached up to 92%). As observed in the 10% canopy openness, shade intolerant native species are negatively impacted by high shade, while the generalist invasive *T. sebifera* maintained its presence and may flourish after hurricane wind disturbances that widen canopy gaps and increase resource availability. Furthermore, *T. sebifera* can withstand storm surge flooding up to three weeks and have the ability to recover after the recession of salinity stress (Howard 2012). Anticipated intensification of tropical storms and storm surge flooding would further facilitate recruitment and regeneration of invasive *T. sebifera*, potentially by increasing light and nutrient availability and reducing competition from some flood intolerant native species (Howard 2012 and references therein).

In conclusion, I found that native and invasive species' performances were increased under simulated canopy openings (a key component of hurricane disturbances). In contrast, effects of simulated storm surge were varied with simulated forest conditions and canopy openness. These results provide an important example of how the population growth, abundance, and persistence of native and invasive species can become altered following hurricane disturbances. Plant species responses to hurricane disturbances will change continuously at different areas over time, ultimately affecting and/or altering future structure and composition of hurricane affected forests across the landscape (Shiels et al. 2010).

Table 4.1. Effects of simulated storm surge on proportion of seedlings mortality of species selected in this study under different simulated canopy openness (30% and 100%) in simulated WPF conditions, and 10% and 70% canopy openness in simulated MHF conditions.

WPF conditions				MHF conditions		
Species	Simulated canopy openness	Storm-Surge	Proportion of mortality	Simulated canopy openness	Storm-Surge	Proportion of mortality
B. halimifolia	30%	Unsurged	0.8	10%	Unsurged	1
		Surged	0.2		Surged	1
	100%	Unsurged	0	70%	Unsurged	0
		Surged	0.2		Surged	0.2
M. cerifera	30%	Unsurged	0.6	10%	Unsurged	1
		Surged	0.8		Surged	1
	100%	Unsurged	0	70%	Unsurged	0
		Surged	0.4		Surged	0.6
T. sebifera	30%	Unsurged	0.4	10%	Unsurged	0.4
		Surged	0.8		Surged	1
	100%	Unsurged	0	70%	Unsurged	0
		Surged	0.4		Surged	0.4

Table 4.2. Summary of repeated-measure analysis of variance test of effects of simulated storm surge, time and their interactions on the relative growth rate of seedlings of species selected in this study under different simulated canopy openness (30% and 100%) in simulated WPF conditions, and 10% and 70% canopy openness in simulated MHF conditions. Significant results are indicated in bold.

	WPF Conditions								MHF Conditions							
Source of variation	30% canopy openness				100% canopy openness				10% canopy openness				70% canopy openness			
	Num DF	Den DF	F	*P*	Num DF	Den DF	F	*P*	Num DF	Den DF	F	*P*	Num DF	Den DF	F	*P*
B. halimifolia																
Storm surge	1	8	0.0	0.98	1	8	0.77	0.40	-	-	-	-	1	8	0.01	0.91
Month	**8**	**39**	**25.82**	**<.001**	**8**	**57**	**21.86**	**<.0001**	-	-	-	-	**8**	**59**	**7.78**	**<0.001**
Storm surge* Month	**8**	**39**	**2.57**	**0.023**	8	57	1.33	0.24	-	-	-	-	8	59	1.58	0.15
Morella cerifera																
Storm surge	1	8	0.75	0.41	1	8	2.68	0.14	-	-	-	-	1	8	0.03	0.87
Month	**8**	**24**	**6.14**	**0.002**	**8**	**52**	**12.08**	**<.0001**	-	-	-	-	**8**	**45**	**69.83**	**<.0001**
Storm surge* Month	8	24	0.38	0.92	**8**	**52**	**4.29**	**0.005**	-	-	-	-	8	45	0.87	0.55
Triadica sebifera																
Storm surge	1	8	0.32	0.58	1	8	4.22	0.074	1	8	0.10	0.76	1	8	4.17	0.07
Month	**8**	**38**	**3.11**	**0.008**	**8**	**54**	**13.40**	**<.0001**	**8**	**27**	**11.90**	**<.001**	**8**	**50**	**10.68**	**<.0001**
Storm surge* Month	8	38	1.38	0.23	**8**	**54**	**3.75**	**0.001**	2	27	1.28	0.29	**8**	**50**	**2.68**	**0.015**

Table 4.3. One-way ANOVA table of the effect of simulated storm surge on total biomass of species selected in this study under different simulated canopy openness (30% and 100%) in simulated WPF conditions, and 10% and 70% canopy openness in simulated MHF conditions. Significant results are indicated in bold.

Source of variation	WPF conditions						MHF conditions					
	30% canopy openness			100% canopy openness			10% canopy openness			70% canopy openness		
	F	F	P	DF	F	P	DF	F	P	DF	F	P
B. halimifolia												
Storm surge	1	0.45	0.55	1	0.96	0.35	-	-	-	1	0.59	0.46
Total	4			8			-			8		
M. cerifera												
Storm surge	1	0.10	0.80	1	0.31	0.59	-	-	-	1	4.76	0.08
Total	2			7			-			6		
T. sebifera												
Storm surge	1	0.14	0.74	1	0.83	0.39	-	-	-	1	0.52	0.50
Total	3			7			-			6		

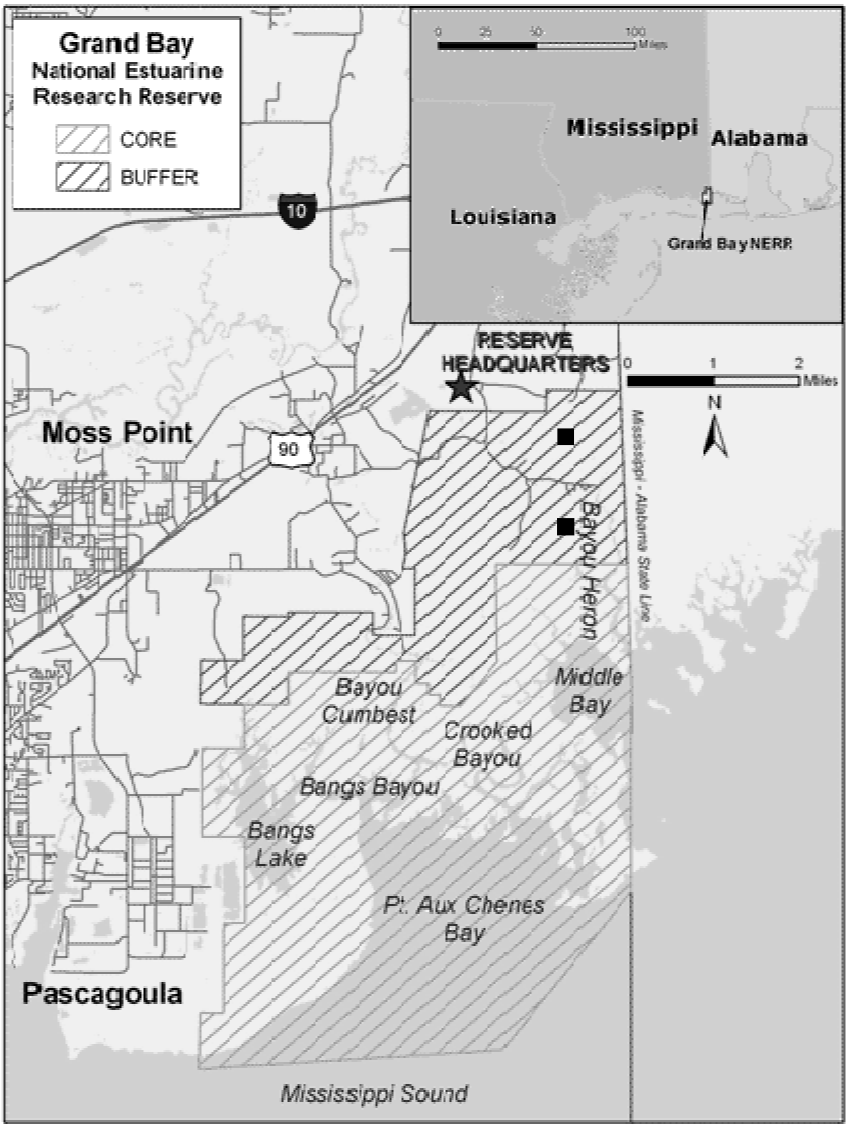

Figure 4.1. A location map of forest stands and GBNERR (courtesy of the Mississippi Department of Marine Resources). Symbol (■) represent forest stands, south to north: wet pine forest (WPF), and mixed hardwood forest (MHF), respectively, used to collect soils for the greenhouse experiment.

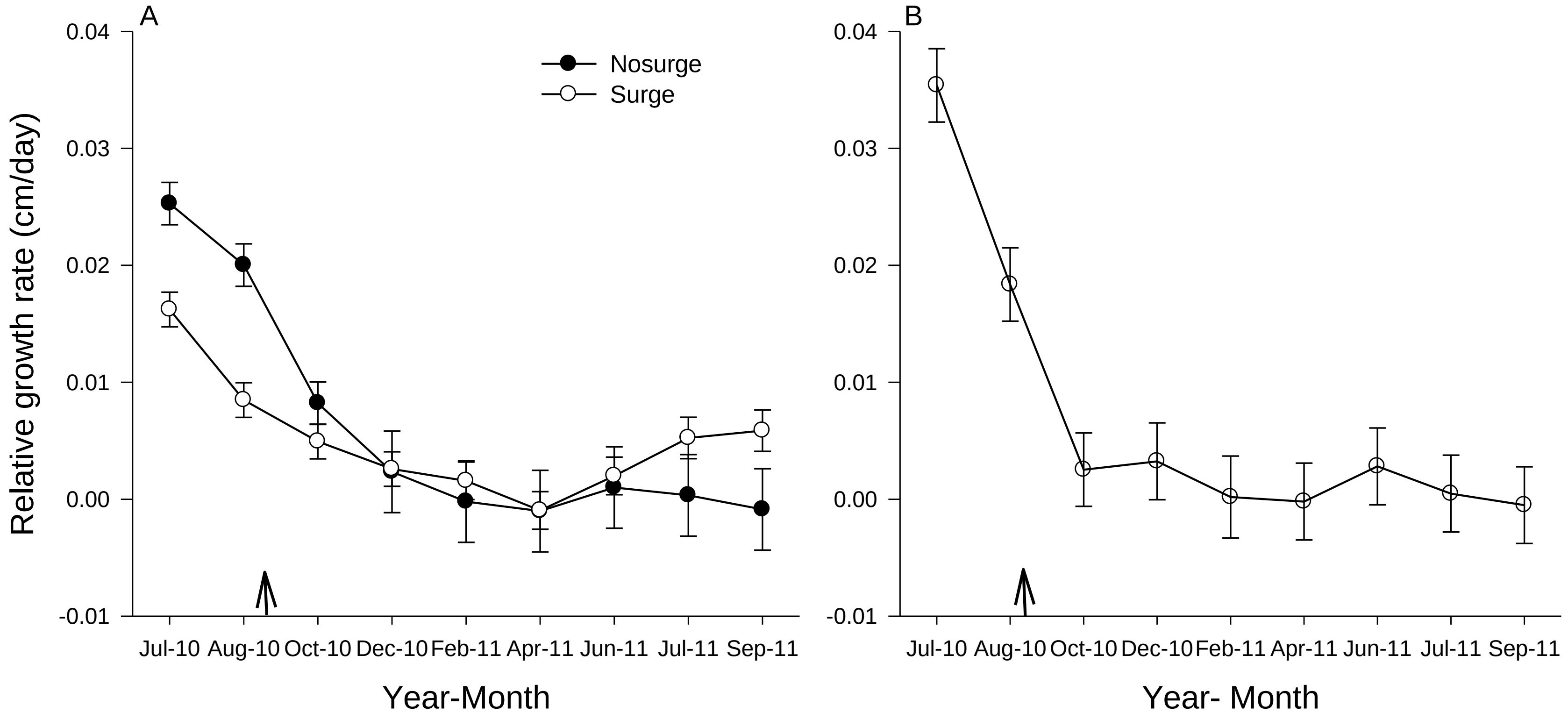

Figure 4.2. Seedlings growth responses (RGR, mean ± SE) to (A) simulated storm surge under simulated 30% canopy openness and (B) over time under simulated 100% canopy openness in simulated WPF conditions for *B. halimifolia* (an arrow denotes the time in which simulated storm surge was applied).

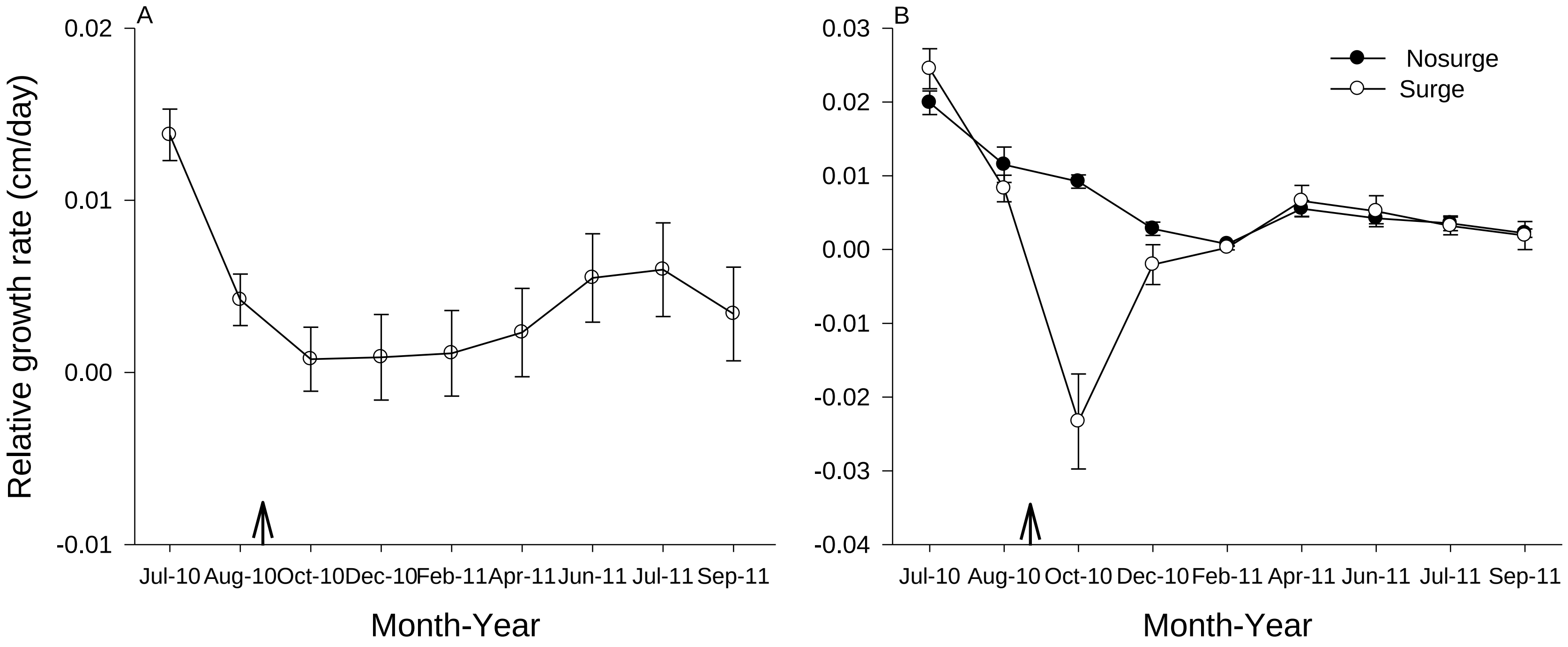

Figure 4.3. Seedlings growth responses (RGR, mean ± SE): (A) over time under simulated 30% canopy openness and to (B) simulated storm surge under simulated 100% canopy openness in simulated WPF conditions for *M. cerifera* (arrow denotes the time in which simulated storm surge was applied).

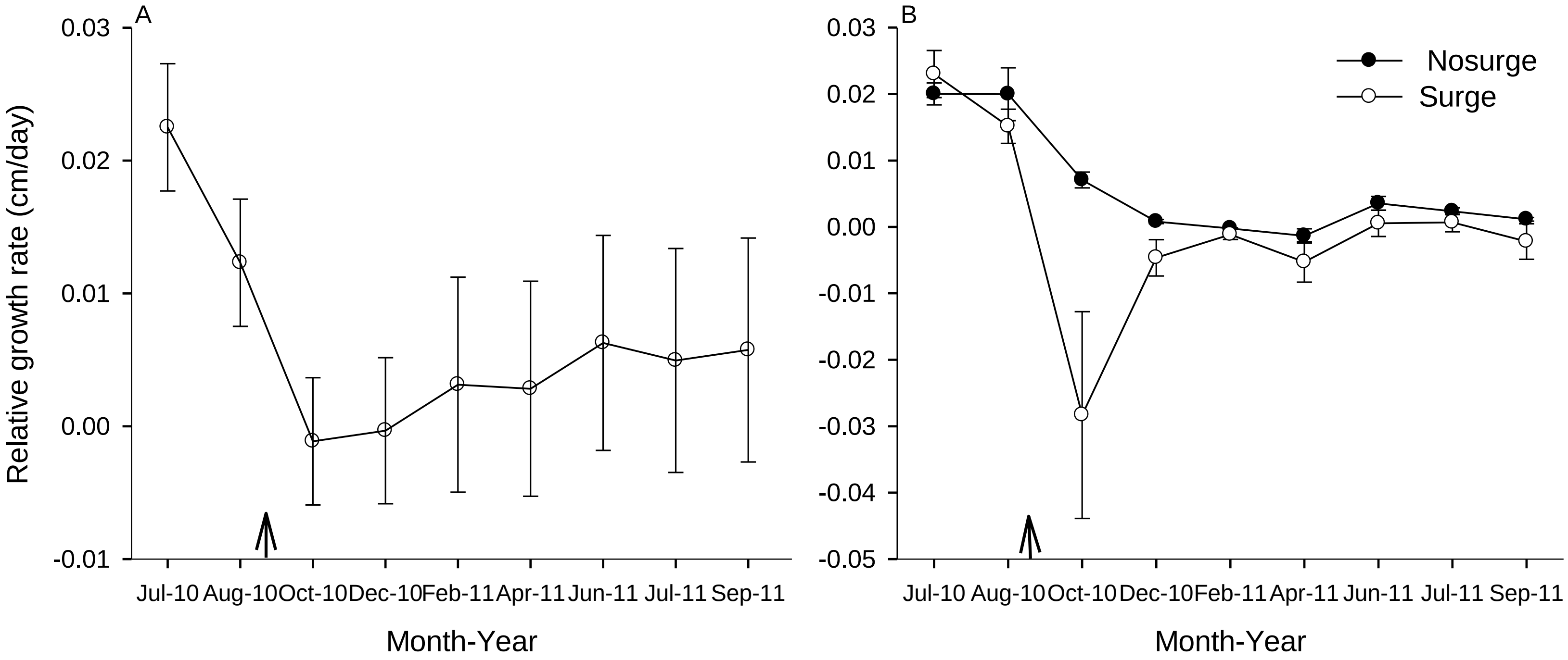

Figure 4.4. Seedlings growth responses (RGR, mean ± SE): (A) over time under simulated 30% canopy openness and to (B) simulated storm surge under simulated 100% canopy openness in simulated WPF conditions for *T. sebifera* (arrow denotes the time in which simulated storm surge was applied).

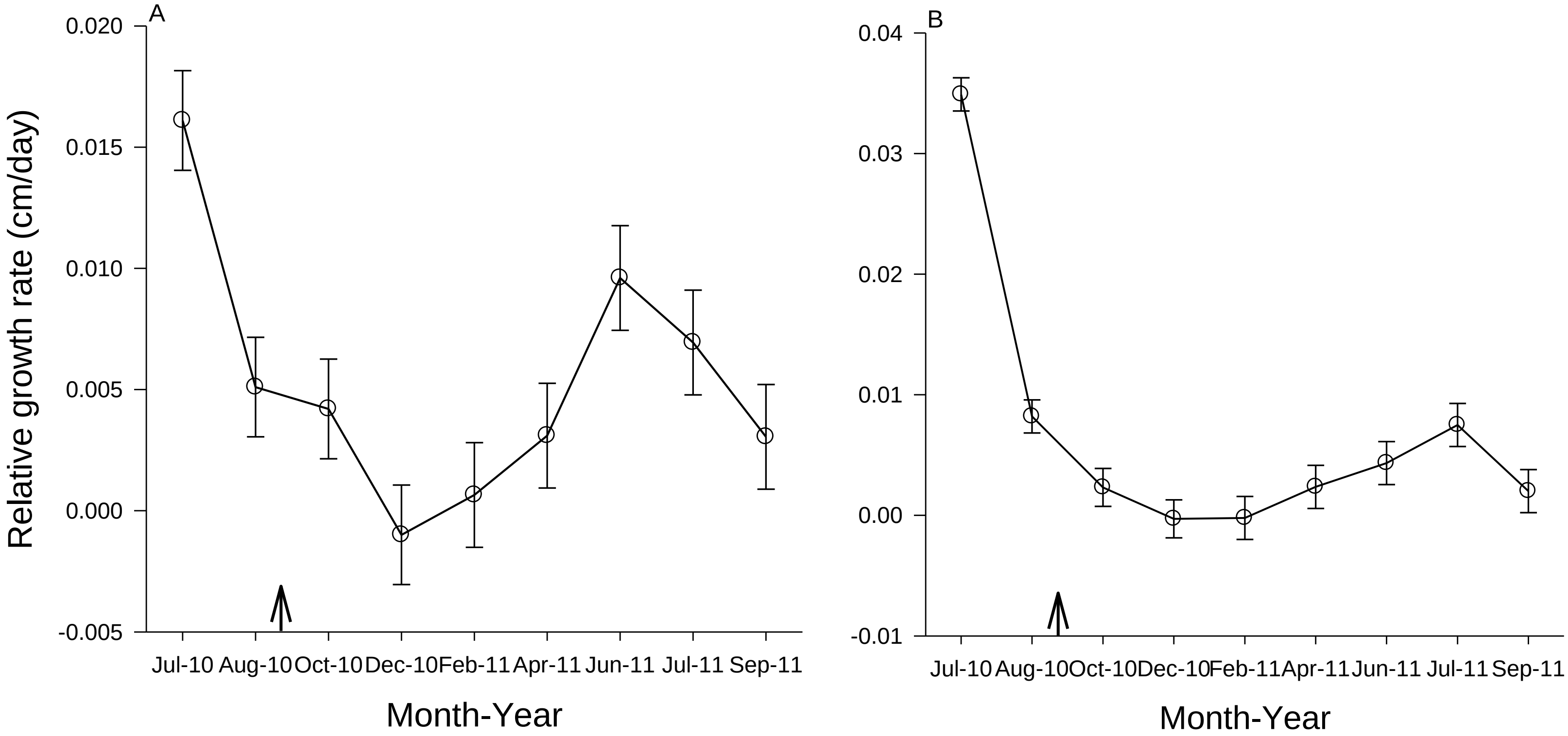

Figure 4.5. Seedlings growth responses (RGR, mean ± SE) over time under simulated 70% canopy openness for (A) *B. halimifolia* and (B) *M. cerifera* in simulated MHF conditions (arrow denotes the time in which simulated storm surge was applied).

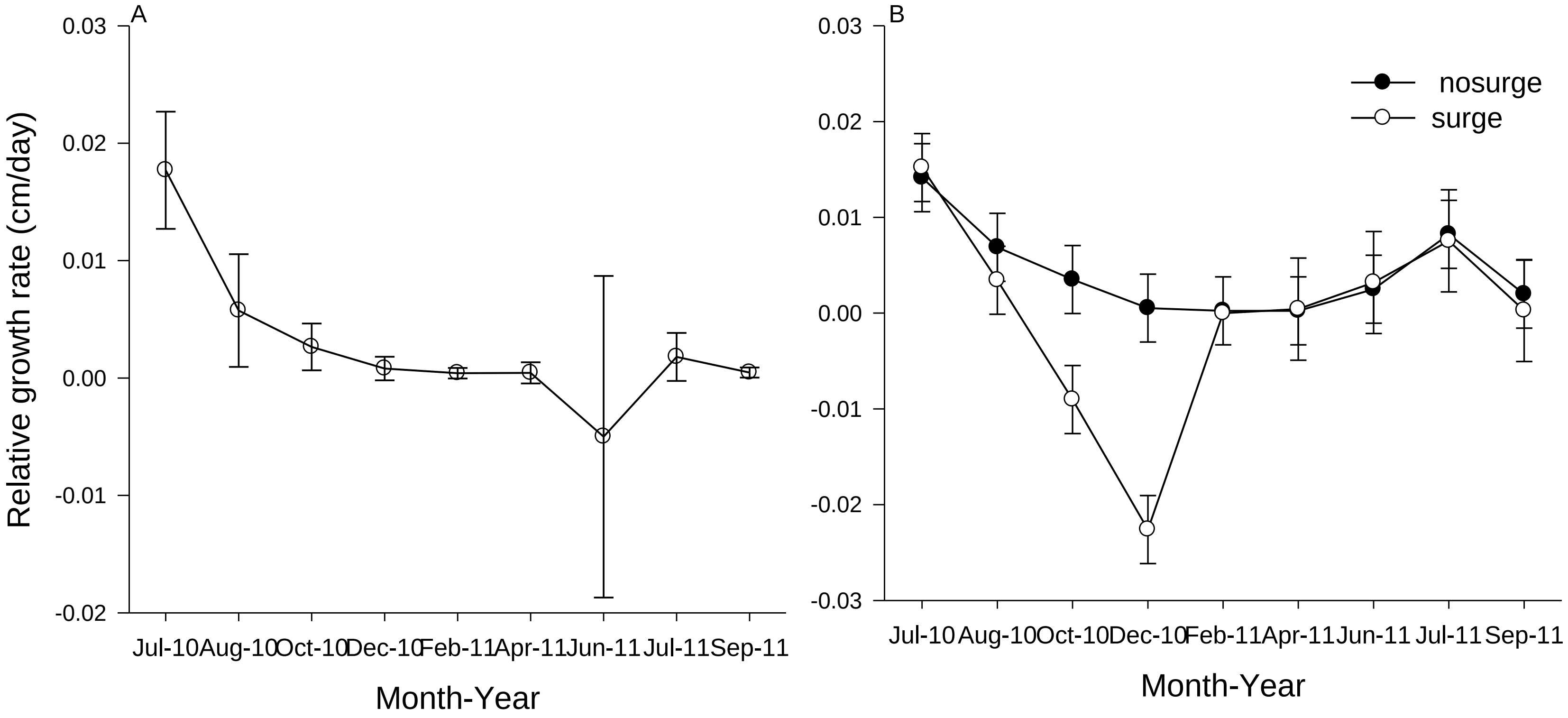

Figure 4.6. Seedlings growth responses (RGR, mean ± SE): (A) over time under simulated 10% canopy openness and to (B) simulated storm surge under simulated 70% canopy openness for *T. sebifera* in simulated MHF conditions (arrow denotes the time in which simulated storm surge was applied).

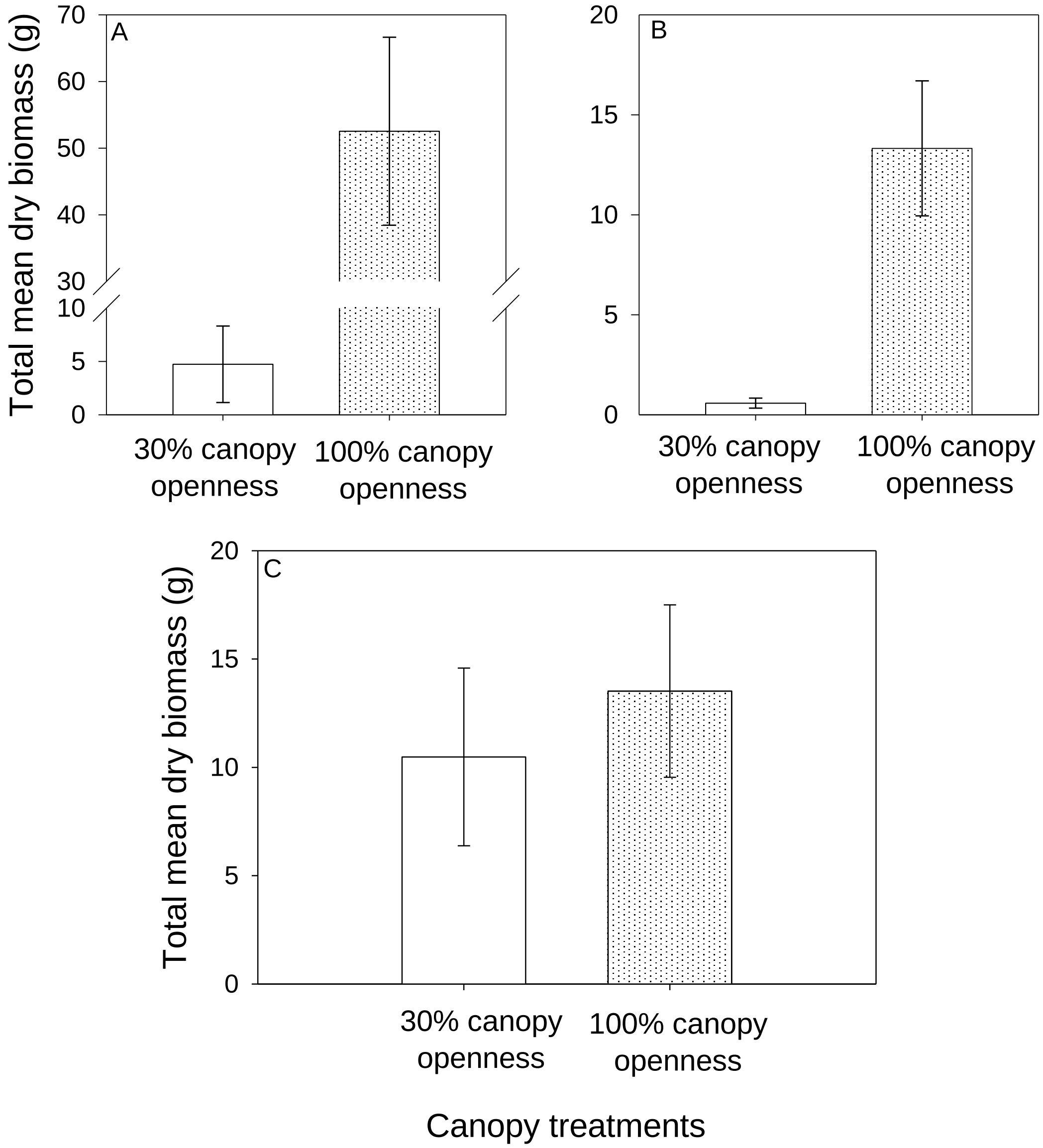

Figure 4.7. Mean (± SE) total dry biomass (shoot + root) for (A) *B. halimifolia* (B) *M. cerifera* and (C) *T. sebifera* after 18 months of growth in the greenhouse under simulated 30% and 100% canopy openness in simulated WPF conditions.

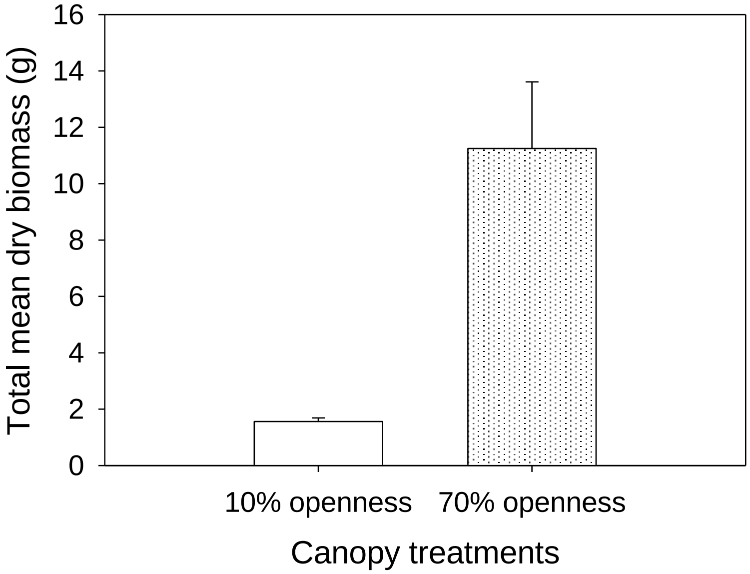

Figure 4.8. Mean (± SE) total dry biomass (shoot + root) for *T. sebifera* after 18 months of growth in the greenhouse under simulated 10% and 70% canopy openness in simulated MHF conditions.

CHAPTER 5

THE ROLE OF VESICULAR ARBUSCULAR MYCORRHIZAE (VAM) ON SUCCESS OF *TRIADICA SEBIFERA* INVASION IN COASTAL TRANSITION ECOSYSTEMS

ABSTRACT

Vesicular arbuscular mycorrhizal (VAM) fungi can have important effects on the structure and diversity of plant communities, in part through promoting exotic plant invasions. Once established, invasive plants can modify the soil-microbial community in directions that benefit themselves. The study of interactions between native plant species, invasive plant species, and soil microorganisms (e.g., VAM) is important to improve our basic understanding of the invasion processes. In a greenhouse experiment, I compared the degree of VAM colonization of the invasive *Triadica sebifera* and two native, co-occurring woody species *Baccharis halimifolia* and *Morella cerifera* and compared their growth in active versus fungicide treated field soils. The invasive *T. sebifera* showed significantly higher VAM colonization in active soils compared to native species and also exhibited significantly higher growth in active soil compared to fungicide treated soil. In a companion field experiment, I tested the effect of VAM on establishment of these species along a typical coastal transition ecosystem in coastal Mississippi, USA. *Triadica sebifera* had higher VAM colonization compared to native species and also maintained higher total biomass during its first eight months of growth. In a second greenhouse experiment, I assessed potential allelopathic effects of *T. sebifera* on *B. halimifolia* and *M. cerifera* by growing them in soils from *T. sebifera* invaded and uninvaded regions with and without activated carbon. VAM colonization of *B. halimifolia* and *M. cerifera* did not differ between the treatments, and also their growth was not affected by soil sources. Overall, my results suggested that *T. sebifera* grows well in active soil and higher degrees of VAM

colonization may be necessary for the initial establishment of this invader. Results from this study also suggested that *T. sebifera* was not allelopathic and did not interfere with growth of native species. Strong positive benefit from VAM may enhance the establishment and capacity of *T. sebifera* to invade coastal plant communities of the southeastern USA.

INTRODUCTION

Soil biota can have significant effects on plant community structure and functions as they, directly or indirectly, influence plant growth, reproduction, and ecosystem processes (van der Heijden et al. 1998, Wall and Moore 1999, Ehrenfeld et al. 2001, Wardle 2002, Callaway et al. 2004). The effects of soil biota on ecosystem processes are often idiosyncratic and can be positive, negative, or neutral depending on which ecosystem, process, and organisms are being described. Positive feedbacks from soil biota on plant species may occur when rhizospheres are infected by mutualistic mycorrhizal fungi (vesicular arbuscular mycorrhizae, hereafter VAM) (Smith and Read 1997) and nitrogen fixing bacteria (Wall and Moore 1999) as both groups of organisms play supportive roles in transporting limited nutrients to the plants. Negative effects arise as a result of accumulation of pathogens, parasites, and root-feeding invertebrates around the rhizosphere (Packer and Clay 2000, Van der Putten et al. 2001, Wardle et al. 2004) and reduce the fitness of plants by decreasing the uptake capacity of roots and removing nutrients and carbon from plant tissues (Bever 1994, Bever et al. 1997).

Soil biota can play important roles in exotic plant invasions as they sometimes promote invasions (Callaway et al. 2004, Wolfe and Klironomos 2005). In general, invasive species can escape soil-borne pathogens in their introduced regions (Agrawal et al. 2005, Reinhart et al. 2010) and are capable of modifying the soil microbial community in directions that benefit themselves (Wolfe and Klironomos 2005, Zhang et al. 2010). When invasive species do encounter pathogens, pathogenic effects may be minimal compared to the mutualistic benefits they receive from VAM (Klironomos 2002). Consequently, mutualistic benefits from soil microorganisms become an important mechanism for establishment of invasive species. A growing body of recent research continues to demonstrate positive feedbacks from VAM to invasive species, which may result in increased dominance within introduced regions (Marler et al. 1999a, Richardson et al. 2000a, Reinhart and Callaway 2004). Some invasive species are capable of developing beneficial mutualisms with VAM at a higher rate than co-occurring native species (Marler et al. 1999b, Nijjer et al. 2008, Smith et al. 2008). Higher degrees of VAM colonization may help to increase plant access to limiting nutrients (e.g., phosphorus) through enhanced mutualism (Reinhart and Callaway 2006, Sun and He 2010).

Some exotic plants are highly successful in their invaded range because they can suppress their native neighbors through the release of chemical compounds (i.e. allelopathy) into their surrounding (Hierro and Callaway 2003, Callaway and Ridenour 2004). Recently, it has been pointed out that exotic invasive plant species can disrupt mutualistic associations between native plant species and their fungal mutualists (Stinson et al. 2006). Stinson and coauthors experimentally demonstrated that the anti-fungal phytochemicals released from the highly invasive garlic mustard can suppress VAM colonization of native tree seedlings in North America and produce negative effects of native tree recruitment. Furthermore, some invasive species have been shown to suppress native species by stimulating soil-borne pathogens that are lethal to native plant species (Mangla et al. 2008). However, limited information is available on (i) whether the degree of VAM colonization is higher in invasive than co-occurring native species, (ii) if the degree of VAM colonization, if any, enhances establishment and growth of invasive species relative to native species, and (iii) whether invasive species disrupt relationships between native plant species and their mutualists through allelopathy. Understanding interactions between invasive plant species, native plant species, and soil biota can be critical to predicting the trajectories of plant invasions and invaded communities.

I used *Triadica sebifera* (L.) Roxb. as a focal invasive species and *Baccharis halimifolia* L. and *Morella cerifera* (L.) Small as focal native species. I assessed the degree of VAM colonization, VAM dependency, and effect of VAM colonization on the performance of these species. I also examined the potential allelopathic effects of *T. sebifera* on focal native species. *T. sebifera* is a small- to medium-sized subtropical, deciduous, monoecious tree in the family Euphorbiaceae. After its introduction into the United States in the late 18th century (Randall and Marinelli 1996), the species has become a serious threat to many low lying coastal ecosystems of the southeast USA (Bruce et al. 1995, Jubinsky and Anderson 1996). In coastal prairies, *T. sebifera* is causing large-scale ecosystem modifications by displacing native plant species (Bruce et al. 1995, Jubinsky and Anderson 1996). In coastal Mississippi, *T. sebifera* is spreading slowly in low-lying coastal communities, particularly in disturbed areas (Matlack 2002). As a result of strong negative impacts on native communities, The Nature Conservancy has categorized *T. sebifera* as one of the worst exotic plants in the USA (Flack and Furlow 1996). Recent studies indicated that some *T. sebifera* populations of the southeastern USA are mycorrhizal (Nijjer et al. 2008, Kandalepas et al. 2010) with higher degrees of VAM colonization compared to co-occurring resident species. It has also been suggested that the species is allelopathic (Gresham 1994); however, no previous studies have directly tested for *T. sebifera*'s VAM dependency and allelopathic effects within invaded communities.

Baccharis halimifolia is a deciduous understory shrub commonly found growing along the Atlantic and Gulf Coast Plains of North America (Duncan et al. 1957). It grows in moist and highly organic soil (Duncan et al. 1957), and has a moderate level of salinity tolerance (Young et al. 1994). One recent study indicated that some *B. halimifolia* populations of the southeastern USA are mycorrhizal (Kandalepas et al. 2010). *M. cerifera* is an evergreen nitrogen-fixing early successional shrub native to wetlands of the Atlantic and Gulf Coastal Plains. It is moderately salt tolerant (Young et al. 1995, Tolliver et al. 1997), and some *M. cerifera* populations along the Atlantic coast are known to be mycorrhizal (Semones and Young 1995).

In a first greenhouse experiment, I assessed the degree of VAM colonization in native *B. halimifolia* and *M. cerifera*, and invasive *T. sebifera*. In the same experiment, I examined the effect of VAM colonization on overall growth of these species. I hypothesized that native and invasive species would have different levels of VAM colonization and that seedlings growing in VAM unsuppressed soil would exhibit higher total biomass. In a companion field experiment, I also assessed the degree of VAM colonization among these species at different seasons. In the same experiment, I examined the role of VAM colonization on establishment and growth of these species along a typical coastal transition ecosystem. I hypothesized that the native and invasive species would have different levels of VAM colonization and higher degree of VAM colonization would positively correlate with total biomass. In a second greenhouse experiment, I explored the potential allelopathic effects of *T. sebifera* on the growth of two native species *B. halimifolia* and *M. cerifera*. I hypothesized that growth of native species would be inhibited in soils with *T. sebifera* invasion relative to growth in uninvaded soils.

MATERIALS AND METHODS

Study System-

The study system was a coastal transition ecosystem at Grand Bay National Estuarine Research Reserve (GBNERR) in coastal Mississippi, USA (30° 21.551'N, 88° 25.202'W) (Figure 5.1). The reserve, located in the low-lying Gulf Coastal Plain, is relatively undisturbed (Hilbert 2006) and contains one of the most biologically diverse and productive estuarine ecosystems on the Gulf of Mexico (NOAA 2007). The reserve includes several plant assemblages that are arrayed along the marine-terrestrial transition and common in Coastal Plain ecosystems: salt marsh, brackish marsh, freshwater marsh, maritime pine forests, wet pine forests, mixed pine hardwood forests, pine flatwoods (Battaglia et al. 2012). The plant communities at GBNERR were historically maintained by natural disturbances (such as fire and tropical storms). The focal plant species occur within some of the forested wetlands of the reserve. For this study, I selected three common vegetation zones, namely brackish marsh, wet pine forest (WPF), and pine flatwoods (PFW) (Figure 5.1).

Before initiating experiments, I checked the status of VAM colonization in those focal species. Although recent field assessments from coastal Louisiana suggested that *B. halimifolia* and *T. sebifera* are mycorrhizal (Kandalepas et al. 2010), there was no information available whether actinorhizal *M. cerifera* is mycorrhizal in low-lying coastal forested wetlands along northern Gulf of Mexico. I collected root samples of all the species from different habitats along the coastal transitions in GBNERR, processed, and examined for VAM colonization. I found all the species were mycorrhizal under natural field conditions.

Experiment 1: Mycorrhizae Colonization and Dependency Experiment-

A greenhouse experiment was conducted to determine the degree of VAM colonization among focal plant species and their VAM dependency on growth by growing the focal species in VAM suppressed and active soil. To initiate the experiment, first a stock of seedlings of all the species were prepared by germinating them in sterilized peat soil. In December 2009, I collected seeds of *B. halimifolia*, *T. sebifera*, and *M. cerifera* plants from multiple locations within GBNERR and cold-stratified these seeds at 4 $^{\circ}$C (Baskin and Baskin 1998) for two months. In late February 2010, seeds were sown and germinated in 52 cm x 26 cm size flats, half-filled with peat-moss soil sterilized by autoclaving at 121 °C for 45 minutes. Before being sown, seeds were washed thoroughly in 10% bleach to remove any surface infection that may have existed on the seeds.

I used field soil collected from three distinct habitats (with respect to distance to the shoreline, vegetation, hydrology, and salinity) at the GBNERR (Figure 5.1). By collecting soils from different habitats I represented the natural range of variability in habitat conditions. Since a typical coastal transition represents a wide range of soil conditions, salinity, and hydrologic gradients, I expect that mycorrhizal abundance and their associations with plant species may also vary along these gradients. The most seaward habitat was brackish marsh ~ 4 km inland from the edge of the coast. The intermediate habitat was considered wet pine forest (WPF) located ~ 6 km inland from the edge of the coast and common in the GBNERR. The most inland habitat, considered wet pine flatwoods (PFW), was ~ 11 km inland from the coast. From each habitat, soils were collected from the top 15 cm at five randomly chosen locations and combined to form one composite sample for each habitat. Before potting, I cleared litter, roots, and any other plant parts present in the soil by passing each composite soil through a 1 cm^2 sieve. The soil was placed into a total of 24 experimental pots (top diameter 12.7 cm x 24.1 cm tall, Stuewe & Sons, Inc. Oregon, USA) assigned for each soil source.

In mid-June 2010 I planted one seedling of a species in each pot. Each pot was replicated four times for each species and soil source. Since the soil utilized in the experiment was taken from the field, I did not inoculate soil with VAM. To apply a non VAM treatment, half of the pots dedicated to each species were treated with fungicide (Brand name: Allban® Flo, OH, USA). All the pots were randomly arranged in the greenhouse and rotated every other week until the experiment was completed in late October 2010. This experiment was complete factorial design yielded a total of 24 pots for each species (3 soil sources × 2 fungicide levels × 4 replicates = 24). I applied Allban to non-VAM treatment immediately after seedlings were transplanted and reapplied it every three weeks. Allban application was intended to reduce VAM colonization and activity. Allban® Flo contains the active ingredient, Thiophanate-Methyl and was advertised as a direct Benomyl substitute (The Scotts Company LLC), as Benomyl was no longer available in the market. The use of Allban® Flo to suppress fungal infections in some crop plants have recently been reported (Daughtrey 2011). For each application, 750 ml of Allban® Flo solution (prepared by adding 1.55 ml of active ingredient into 1 liter of de-ionized water) was applied to each non-VAM treatment. Transplanted seedlings were watered with normal tap water to saturation twice per week in order to maintain field capacity soil moisture.

After 122 days of growth, seedlings were harvested and separated into shoots and roots. Roots were washed with de-ionized water and small subsamples of fine roots from each seedling were separated and processed for VAM colonization. Shoots and roots were dried to a constant weight at 55 °C for 7 days and weighed to determine dry biomass. Small root samples, separated to estimate VAM colonization, were stored in 50% ethanol until roots were processed. Roots stored in 50% ethanol were cut into 1-cm long fragments and cleared in 10% potassium hydroxide (KOH) by placing root fragments in KOH solution overnight. Cleared roots were then rinsed with deionized water and acidified in 1% HCl for 15 min. Acidified roots were stained in Trypan blue (1:2:2 lactic acid, glycerol, deionized water with 0.6 g Trypan blue added per liter) for about 4-5 hours. Nine randomly selected stained fragments were mounted on three microscope slides (each slide had three root fragments) and examined for VAM colonization. Each mounted root was observed under a compound microscope (Olympus-BH-2) using a 40 × objective lens and checked for arbuscules, vesicles, and hyphae of VAM. All the lab procedures for processing root samples to estimate VAM colonization were slightly modified from Reinhardt and Miller (1990) and McGonigle et al. (1990), and percent VAM colonization was determined using the magnified intersection method (McGonigle et al. 1990). Since seedlings of all the species transplanted to the soil from brackish marsh did not survive, I discarded that habitat from the experiment.

Experiment 2: Field Experiment-

To investigate the degree of VAM colonization over time and the effects of colonization on establishment, growth, and performance of focal species in natural field conditions, I conducted a 15 month field experiment. I selected the same three habitats at GBNERER used as soil sources for the first experiment (Figure 5.1). On June 14, 2010, I randomly placed a group of nine pots (3 replicate pots of 25 cm top diameter × 25 cm height for each species) in each of the three habitats. A pot-size hole was made on the ground by excavating soil using a shovel. The excavated soil was then put into the pot and the pot was positioned back into the same hole. The level of the soil in the pot coincided with the outside ground level. Pots had three equal-sized drain holes at the bottom and made additional five drain holes each at 10 cm and 20 cm from the bottom in order to encourage water flow through the soil in the pots. I utilized the stock of seedlings prepared for experiment 1, and transplanted five seedlings of each species in each designated pot and seedlings were numbered 1-5. Seedlings were watered to saturation three times (first, third and fifth day) at the beginning of the experiment and then allowed to grow under natural field conditions for 15 months. Survival of transplanted seedlings was monitored for the first five days and after one month. I noticed all transplanted seedlings at the WPF and PFW habitats were intact but, as in the first experiment, all the seedlings in the brackish marsh had died. Thus, I omitted that habitat from the experiment.

A total of three seasonal harvests were made for all the species from both habitats by randomly selecting a seedling from each pot. Before harvest, I made notes on the status (e.g., browsing and herbivory) of growing seedlings. During the first two harvests, widespread browsing (mainly apical portion of the seedling) was observed for all the species, and therefore, I did not incorporate relative growth rate into the analysis. The first, second, and third harvests were made in September 2010 (after approximately 75 days of growth), March 2011 (after approximately 240 days of growth), and September 2011 (after approximately 450 days of growth), respectively. Coming to the end of the experiment period, none of the *B. halimifolia* and *T. sebifera* and only one of *M. cerifera* seedlings had survived at PFW, so the September 2011 harvest represented only WPF habitat. Under normal conditions, PFW is drier than WPF because the former occupies the more inland location along the coastal transition of the two. I suspect that a long drought during late spring and early summer of 2011 (Figure 5.2) likely produced those results. Previously it was reported that severe droughts can substantially decrease survival of young *T. sebifera* seedlings (Bower et al. 2009).

For each harvest, seedlings were separated into shoots and roots. Roots were washed thoroughly in tap water and small subsamples of fine roots were separated to assess status of VAM colonization among species. Fine roots were washed in de-ionized water and stored in 50% ethanol until the roots were processed. The protocol of processing roots to estimate VAM colonization was the same as described for experiment 1. Shoots and roots were dried to a constant weight at 55 °C for 7 days and weighed to determine dry biomass.

Experiment 3. Allelopathy Experiment-

To investigate the potential allelopathic effects of *T. sebifera* on native species, *B. halimifolia* and *M. cerifera*, I grew *B. halimifolia* and *M. cerifera* seedlings in soil with and without the history of *T. sebifera* invasion. I collected soil from *T. sebifera* invaded and *T. sebifera* free (greater than 50 m away from the nearest *T. sebifera* tree) areas at GBNERR. At each area, soil was collected from 10 randomly chosen locations and the soil collection protocol was the same as described for experiment 1. *Triadica sebifera* invaded and uninvaded soils were pooled separately in the lab and sieved (1 cm^2) to remove coarse roots, debris, and litter. Two separate experiments (each for *B. halimifolia* and *M. cerifera*) were conducted in a greenhouse setting. For each species, a total of 20 pots (the same pots used in experiment 1) were filled with sieved soil. For each species, one fourth of the pots (5) received soil collected from *T. sebifera* invaded soil mixed with finely ground activated carbon (Carbochem Inc., PA, USA) (20 ml per liter of soil) while one other fourth of the pots (5) received soil collected from the *T. sebifera* free area mixed with the same amount (i.e. 20 ml per liter of soil) of activated carbon. Mixing activated carbon into the soil reduces the allelopathic effects of organic root exudates because activated carbon has a high affinity to allopathic organic compounds but at the same time has low affinity to inorganic compounds in nutrient solutions (Callaway and Aschehoug 2000).

Among the remaining 10 pots for each species, 5 received soil from *T. sebifera* invaded areas and 5 received soil from *T. sebifera* uninvaded areas with no activated carbon added.

In the 2nd week of June 2011 I planted two seedlings (prepared following the same method as described for experiment 1) of one of the two native species (*B. halimifolia* or *M. cerifera*) to each pot in a complete factorial design (2 soil sources × 2 carbon × 5 replicates of each treatment combination = 20 pots for each species). Pots were arranged randomly on the greenhouse bench and re-randomized often. Plants were watered to saturation twice a week (however, during extremely hot weeks, watered every other day). After 4 months of growth, shoots and roots were harvested. Roots were washed thoroughly in tap water. Shoots and washed roots were dried at a constant weight at 55 °C for a week and weighed to determine biomass. Small subsamples of fine roots from seedlings of each pot were separated and processed for VAM colonization as described for experiment 1. I examined the degree of VAM colonization in roots of *B. halimifolia* and *M. cerifera* seedlings, grown in soils with and without a history of *T. sebifera* invasion. Seedlings from each pot were combined and measured as one biomass for both shoots and roots. Before analysis, I combined both shoot and root biomass and expressed as total biomass.

Statistical Analyses-

In a first greenhouse experiment and using unmanipulated soils only, the degree of VAM colonization among species (3 species) and across soil source (2 levels) was compared using two-way ANOVA. Percent colonization data were natural log-transformed to meet assumption of normality prior to analysis. I combined dry shoot and root biomass into total dry biomass for all species. Total biomass data were analyzed for each species separately using two-way ANOVA for the fixed effects of soil source (2 levels) and fungicide (2 levels) and their interaction. Total biomass data for *B. halimifolia* were natural-log-transformed for normality, while total biomass data for *M. cerifera* and *T. sebifera* were normally distributed and did not require transformation. I also performed a linear regression to determine the relationship between percent VAM colonization and total biomass for each species.

Data from the field experiment for percentage VAM colonization and total dry biomass (root + shoot) during first two harvests were compared among the species (3 species) and across the habitats (2 levels) using two-way ANOVA for the fixed effects of habitat and species and their interaction. However, all species were dead before the third harvest at PFW and comparisons for VAM colonization and total dry biomass among species were made for WPF only by using one-way ANOVA. Percent VAM colonization and total biomass data from the first harvest were normally distributed and did not require transformation, while same data from the second harvest were natural-log-transformed for normality. For the third harvest, the percent of VAM colonization and total dry biomass data were normally distributed and did not require transformation.

Total dry biomass and percent VAM colonization data for each species from experiment 3 were analyzed using two-way ANOVA for the fixed effects of soil source (2 levels) and presence of activated carbon (2 levels) and their interactions. Data were normally distributed and did not require transformation. All data were analyzed using SAS version 9.2 (SAS 2008) and the alpha level used for detecting significant difference was $\alpha = 0.05$.

RESULTS

Experiment 1. Mycorrhizae Dependency Experiment-

I observed VAM colonization (arbuscules, hyphae, and vesicles) in roots of all three plant species grown in fungicide treated and control field soils. Fungicide application substantially reduced VAM colonization of roots, regardless of habitats, from an average of 21.71 ± 3.32% to 6.60 ± 1.36% for *B. halimifolia*, from 8.92 ±1.97% to 2.99 ±0.76% for *M. cerifera*, and from 40.38 ± 5.30% to 5.63 ± 1.25% for *T. sebifera* (see Table 5.1 for ANOVA results). In control soils, the degree of VAM colonization varied among species regardless of habitats (two-way ANOVA: main effect; $F_{species} = 18.06$, df = 2,70; $P < 0.0001$). The invasive *T. sebifera* showed a significantly higher degree of VAM colonization compared to native species (Figure 5.3).

Total seedling dry biomass of *B. halimifolia* and *M. cerifera* was not significantly affected by VAM suppression (Figure 5.4A and 5.4B, respectively), habitats, and their interaction ($P > 0.05$; Table 5.2). However, there was a significant positive relationship between total seedling dry biomass and VAM colonization for *B. halimifolia* (Figure 5.5A); but not for *M. cerifera* (Figure 5.5B). In contrast, *T. sebifera* total seedling dry biomass was significantly affected by VAM suppression regardless of habitats (Table 5.2, Figure 5.4C); I also found a significant positive relationship between total dry biomass and VAM colonization (Figure 5.5C).

Experiment 2. Field Experiment-

VAM colonization was observed in all three species at both habitats in all harvests. During the first two harvests, regardless of habitats, there were clear trends in VAM colonization among species and the invasive *T. sebifera* had higher VAM colonization (fall of 2010: main effect, $F_{species} = 43.43$, df = 2,50, $P < 0.001$, Figure 5.6 and spring of 2011: main effect, $F_{species} = 30.45$, df = 2,38, $P < 0.001$, Figure 5.6). Also in the final harvest, *T. sebifera* had higher VAM colonization compared to native species at WPF (fall of 2011: one-way ANOVA; species effect, $F_{species} = 30.45$, df = 2, 18, $P < 0.001$, Figure 5.6).

Total dry biomass was higher for *T. sebifera* compared to other species after 2 ½ months of growth (fall of 2010) regardless of habitats (Table 5.3, Figure 5.7A). After eight months of growth (spring of 2011), a similar trend was observed with a significant habitat × species interaction (Table 5.3, Figure 5.7B). At the conclusion of the experiment (fall of 2011), there was no significant differences in total dry biomass between species at WPF (fall of 2011: one-way ANOVA; species effects, $F_{species} = 3.123$, df = 2,6, $P = 0.12$).

Experiment 3: Allelopathy Experiment-

Baccharis halimifolia had no significant differences in percent VAM colonization of roots between the treatments (Table 5.4), while the species had significantly higher total dry biomass in activated carbon treatment than without activated carbon treatment ($P = 0.002$; Table 5.5, Figure 5.8A). However, soil source did not have significant effects on total dry biomass ($P > 0.05$; Table 5.5, Figure 5.8B). Similarly, *M. cerifera* had no significant difference in percent VAM colonization of roots ($P > 0.05$; Table 5.4) or total dry biomass between the treatments ($P > 0.05$; Table 5.5).

DISCUSSION

Mutualisms between vesicular arbuscular mycorrhizae (VAM) and plants have important influences on growth, development and dominance of plant species (Hartnett and Wilson 1999) as VAM help plants to access limiting nutrients (e.g., phosphorus and nitrogen) (Allen and Allen 1990). In particular, invasive exotic species may receive greater benefits from VAM colonization then their native counterparts when the invasive species has a higher degree of VAM colonization than its native counterparts (Smith et al. 2008) and can receive positive benefit from VAM due to a higher mutualists:pathogen ratio (Klironomos 2002). The greenhouse (experiment 1) and companion field experiments show that the invasive *T. sebifera* consistently had a greater degree of VAM colonization compared to native species, and this association may help *T. sebifera* tap into limited nutrients more efficiently than native species and increase its invasive capacity.

Results from this study support the prediction that soil biota (e.g., mycorrhizal fungi) may play a facilitative role in establishment and spread of invasive species in some of the invaded communities (Marler et al. 1999a, Richardson et al. 2000a, Callaway et al. 2001, Callaway et al. 2004). Significantly higher dry shoot biomass in active soil and a significant positive correlation between degree of VAM colonization and total shoot dry biomass suggests that *T. sebifera* experiences stronger positive benefit from VAM in coastal Mississippi and an absence or low colonization of VAM may have negative effects on *T. sebifera* invasion. This also suggests that the mutualism with VAM fungi may be essential for the success of *T. sebifera* in introduced ranges. It is predicted that mycorrhizal dependent plant species that are invasive are likely to be flexible, forming mutualistic associations with a diverse array of VAM fungi (Rejmanek 2000, Pringle et al. 2009). Successful invasion of the Galapagos Islands by the obligate arbuscular mycorrhizal *Psidium guajava* was possible because of this species' ability to tap into mycorrhizal networks already present on the Islands (Schmidt and Scow 1986). My results suggest that *T. sebifera* may be flexible as well, with higher degree of VAM colonization compared to its native counterparts, which contributes to its successful invasion of coastal ecosystems of the southeastern USA.

Some invasive species may receive stronger positive benefit from soil biota in their introduced range compared in their native range, enhanced mutualism (Reinhart and Callaway 2006), and that the stronger positive benefit may account for the successful invasion of *T. sebifera* in forests of the southeastern USA, as suggested previously for east Texas. Consistent with Nijjer et al. (2008), I found an increased total biomass of *T. sebifera* seedlings in VAM unsuppressed soils regardless of the soil source and habitat where the *T. sebifera* was grown. Higher degrees of VAM colonization compared to native counterparts likely provide improved benefits to *T. sebifera* in invaded communities. Native species may not necessarily experience the similar mutualistic relations with VAM fungi. VAM often enhance growth and development in plants through increased phosphorus (Smith and Read 1997) and nitrogen uptake (Leigh et al. 2009, Hodge and Fitter 2010). In particular, some invasive species (e.g., *Centaurea maculosa*) efficiently utilize their mycorrhizal fungal hyphae to acquire phosphorus, potentially from their neighboring native plants, and enhance their performance (Zabinski et al. 2002). Although, It is difficult to pinpoint this as the reason for increased performance by *T. sebifera* in this case because I did not assess the phosphorus and nitrogen content in the plants and also grew species individually in pots so each species had its own rhizosphere. However, greater degree of VAM colonization in *T. sebifera* compared to native species likely increases the chances of tapping limiting nutrients from the soil and could increase its overall performance relative to native species.

Higher degree of VAM colonization may provide additional benefits (e.g., protection from diseases and soil borne pathogens (Newsham et al. 1995)) to some invasive species, while native species may not have the same benefits (Klironomos 2002). Some invasive plant species may have escaped their natural enemies (enemy release hypothesis) and soil-borne diseases in their invaded range and obtain net positive benefits from mutualists (Klironomos 2002, Callaway et al. 2004, Reinhart and Callaway 2004, Callaway et al. 2011). Previous studies have shown that release from below-ground enemies, soil-borne pathogens, and colonization of mutualists triggered the invasion and superior performance of the highly invasive *Prunus serotina* in north-western Europe (Reinhart et al. 2003) and *Robinia pseudoacacia* (Callaway et al. 2011) in Europe. Klironomos (2002) found that the majority of invasive species (four out of five) experienced significant positive benifit from soil biota but were not affeced by soil-borne pathogens, whereas all of the rare native species experienced negative effects driven by soil-borne pathogens. In this study, however, it is not known whether the release from soil borne pathogen increased the overall performace of *T. sebifera* and whether a higher degree of VAM colonization protect *T. sebifera* roots from diseases and pathogens. Further studies examining these important mechanisms, using comparative biogeographical experiments (i.e., plant performance on native and foreign soil), will help to understand the cause of the effect (Brinkman et al. 2010).

Consistent with recent research (Conway et al. 2002), my results did not provide evidence for allelopathic effects of *T. sebifera* root exudates on common native species in this coastal transition ecosystem (experiment 3). When native species seedlings were grown in soils from *T. sebifera* invaded and uninvaded areas, the degree of VAM colonization in the roots and total dry biomass were not different between soil sources. Here, *T. sebifera* neither suppressed the mutualism between native plant species and VAM nor had negative effects on native species growth. These results suggest that *T. sebifera* does not disrupt the mutualism between native species and their VAM mutualists in these coastal wetland ecosystems. My results are in contrast with several of the previous studies that reported allelopathic effects of invasive plant species on native counterparts through reducing nutrient uptake (Callaway and Aschehoug 2000), interfering with mutualisms between native plant species and their mutualists (i.e., mycorrhizal fungi) (Roberts and Anderson 2001, Stinson et al. 2006, Wolfe et al. 2008), and reducing growth of mycorrhizal fungi (Vogelsang and Bever 2009). Furthermore, my results did not support Gresham's (1994) findings that *T. sebifera* produced negative effects on native *Pinus taeda* (loblolly pine) in the lower Coastal Plain of South Carolina through direct inhibition of seed germination and seedling growth via allochemicals. However, my results agreed with other studies on *T. sebifera* from coastal Texas. These studies did not report allelopathic effects of *T. sebifera* on co-occurring native species' germination and seedling growth (Keay et al. 2000, Conway et al. 2002). Conway et al. (2002) also reported a substantial increase in seed germination and seedling growth of the invader when an aqueous extract of *T. sebifera*'s leaves and litter was applied into its growing medium; and they proposed that self-facilitation might explain successful invasion of *T. sebifera* rather than its allelopathic interference with native plant species survival.

Results from the allelopathy experiment showed that the seedlings of *B. halimifolia* growing in soil mixed with activated carbon produced significantly higher total biomass compared with seedlings growing without activated carbon. Previous studies showed that activated carbon can increase aboveground biomass through increased nutrient availability (e.g., nitrogen) (Lau et al. 2008), and at the same time reduced VAM infection to the plant (Weisshuhn and Prati 2009). Although, I did not find the negative effects of activated carbon on VAM colonization, there was a clear evidence of a direct effect of activated carbon on total dry biomass of *B. halimifolia*. So, activated carbon should be used cautiously when performing allelopathy experiments because activated carbon may influence soil chemistry, interact with VAM, and produced undesired effects (Weisshuhn and Prati 2009).

In conclusion, findings of this research demonstrated that mycorrhizae may play a key role during the invasion processes in some exotic plant species. A higher degree of VAM colonization that may produce stronger positive benefit for the invader (enhanced mutualism) may have facilitative effects on successful invasion of *T. sebifera* in various coastal plant communities of the southeastern USA. Comparative biogeographical approaches, which compare the feedbacks from soil biota to exotics in their introduced and native ranges, are vital to acquire invaluable information on invasion processes and success of exotics in introduced ranges (Hierro et al. 2005). These approaches would further our understanding of the influence of soil biota on *T. sebifera* success as an invader.

Table 5.1. Experiment 1: Summary of two-way ANOVA of effects of fungicide (fungicide and control) and soil source (WPF and PFW) on VAM colonization on *B. halimifolia*, *M. cerifera* and *T. sebifera* seedlings grown in the greenhouse.

Source of variation	Df	Mean square	F	*P*	Df	Mean square	F	*P*	df	Mean square	F	*P*
	B. halimifolia				*M. cerifera*				*T. sebifera*			
Soil source	1	42.25	0.29	0.59	1	4.08	2.79	0.06	1	0.19	0.00	0.98
Fungicide	**1**	**2653.48**	**18.42**	**0.0001**	**1**	**8.16**	**9.39**	**0.003**	**1**	**14181**	**40.68**	**0.0001**
Soil source * Fungicide	1	573.61	3.98	0.060	1	0.12	0.14	0.706	1	120.49	0.35	0.55
Total	44				14				15			

Table 5.2 Experiment1**:** Summary of two-way ANOVA of effects of fungicide (fungicide and control) and soil source (WPF and PFW) on total biomass of *B. halimifolia*, *M. cerifera*, and *T. sebifera* seedlings grown in the greenhouse.

Source of variation	Df	Mean square	F	*P*	Df	Mean square	F	*P*	df	Mean square	F	*P*
	B. halimifolia				*M. cerifera*				*T. sebifera*			
Soil source	1	0.058	0.28	0.60	1	0.052	0.32	0.58	1	0.01	0.31	0.58
Fungicide	1	0.446	2.21	0.16	1	0.059	0.37	0.55	1	**0.38**	**9.19**	**0.01**
Soil source * Fungicide	1	0.043	0.23	0.64	1	0.35	2.21	0.16	1	0.02	0.62	0.48
Total	15				14				15			

Table 5.3. Experiment 2: Summary of two-way ANOVA of effects of habitat (WPF and PFW) and species (*B. halimifolia*, *M. cerifera*, and *T. sebifera*) on total biomass of seedlings grown in the field.

Source of variation	2 ½ months (Sept 2010)			8 months (March 2011)		
	df	*F*	*P*	Df	*F*	*P*
Habitat	1	3.18	0.102	1	8.20	0.011
Species	2	24.03	<0.0001	2	18.29	0.002
Habitat *Species	2	0.70	0.518	2	4.80	0.042
Total	16			13		

Table 5.4. Experiment 3: Summary of two-way ANOVA of effect of soil source (TR, *T. sebifera* invaded or un-invaded) and carbon (activated carbon or nocarbon) on VAM colonization of root's of *B. halimifolia* and *M. cerifera* grown in the greenhouse.

Source of variation	*B. halimifolia*			*M. cerifera*		
	df	F	*P*	Df	F	*P*
TR	1	0.81	0.37	1	3.30	0.078
Carbon	1	2.86	0.09	1	3.52	0.069
TR*Carbon	1	0.2	0.65	1	1.70	0.20
Error	36			34		

Table 5.5. Experiment 3: Summary of two-way ANOVA of effect of soil source (TR, *T. sebifera* invaded or un-invaded) and carbon (activated carbon or nocarbon) on total biomass of seedlings of *B. halimifolia* and *M. cerifera* grown in the greenhouse.

Source	Total biomass			Total biomass		
Source	df	F	*P*	df	F	*P*
B. halimifolia				*M. cerifera*		
TR	1	0.01	0.93	1	0.96	0.34
Carbon	1	13.14	0.002	1	1.98	0.17
TR× Carbon	1	0.42	0.52	1	0.24	0.63
Error	19			19		

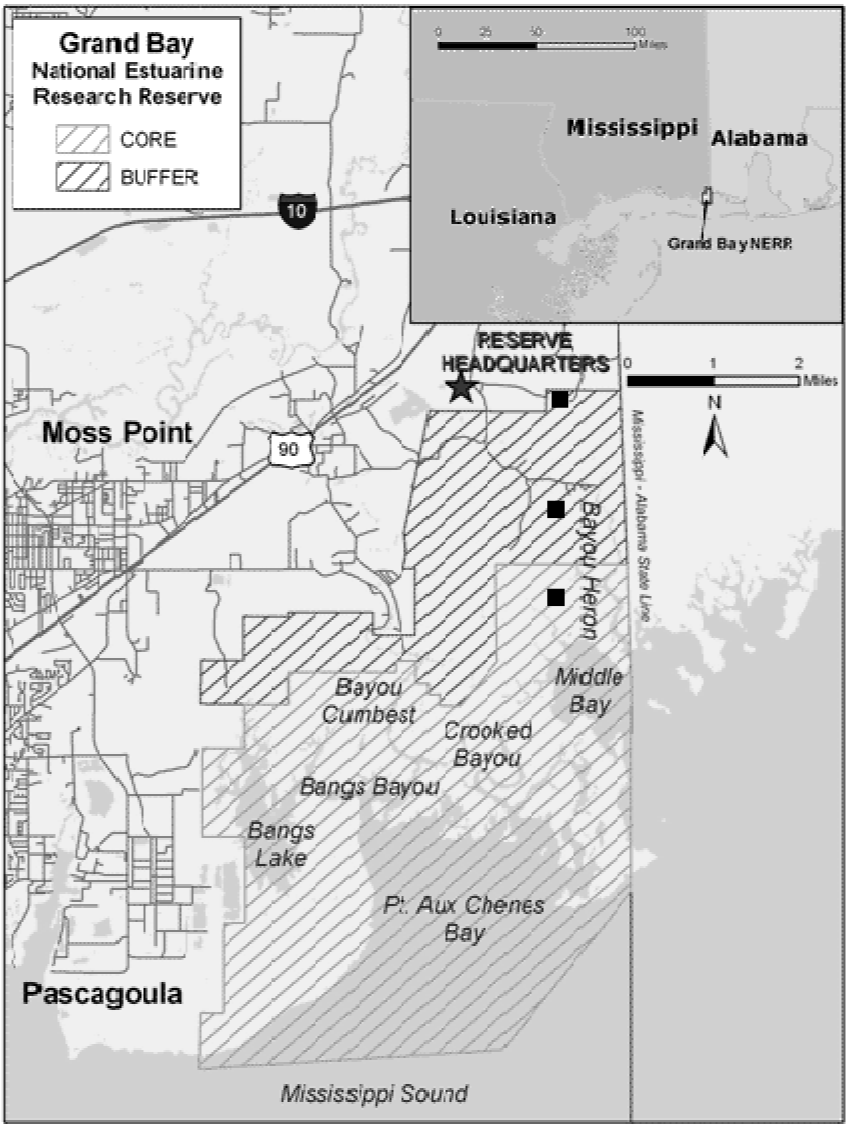

Figure 5.1. Location map of study area (courtesy from Grand Bay National Estuarine Research Reserve. Symbol (■) represent three different habitats (Brackish marsh (BM), wet pine forest (WPF), and pine faltwoods (PFW), respectively from south to north) along the the coastal transition from where soil samples were collected for experiment 1 and same habitats were used for field experiment.

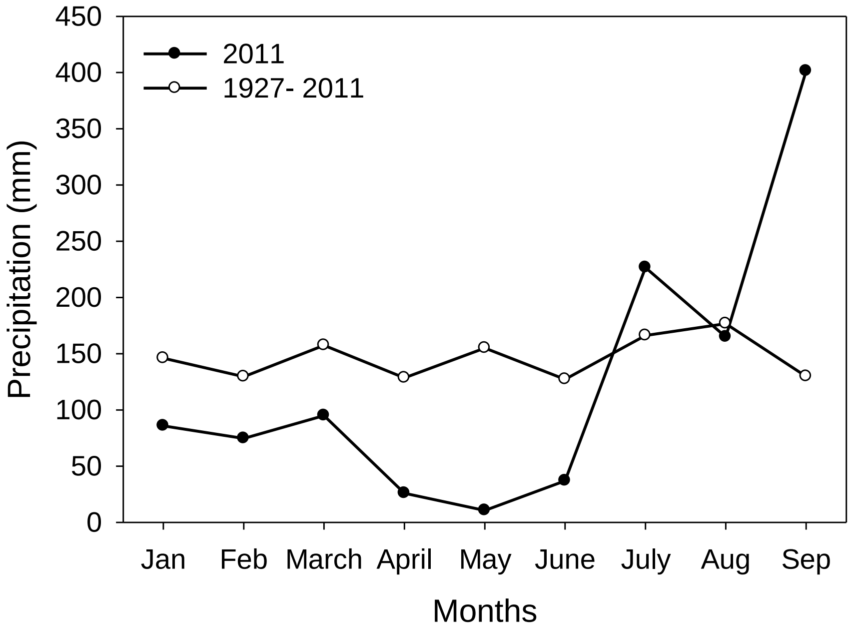

Figure 5.2. Precipitation data for Mobile Alabama near experimental site, representing total monthly rainfall for 2011 and average monthly rainfall for 1927 to 2011 (source: http://www.srh.noaa.gov/mob/?n=climate).

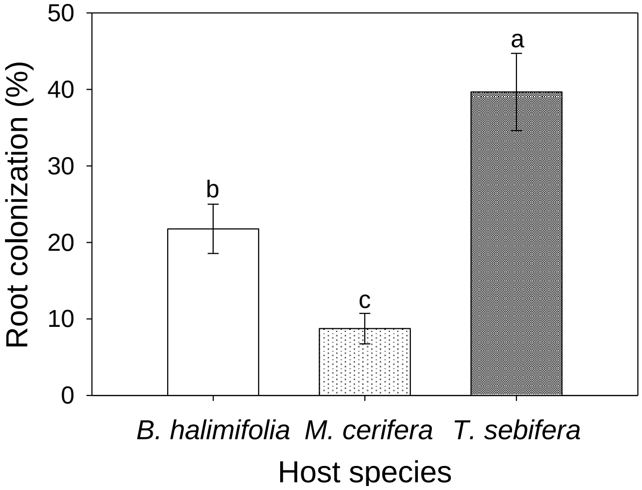

Figure 5.3. Experiment 1: The percentage of VAM colonization (means ± SE) on three species grown in controlled (no-fungicide) field soils collected from two different habitats (WPF and PFW) of coastal Mississippi. Different letters indicate significantly different means ($\alpha = 0.05$).

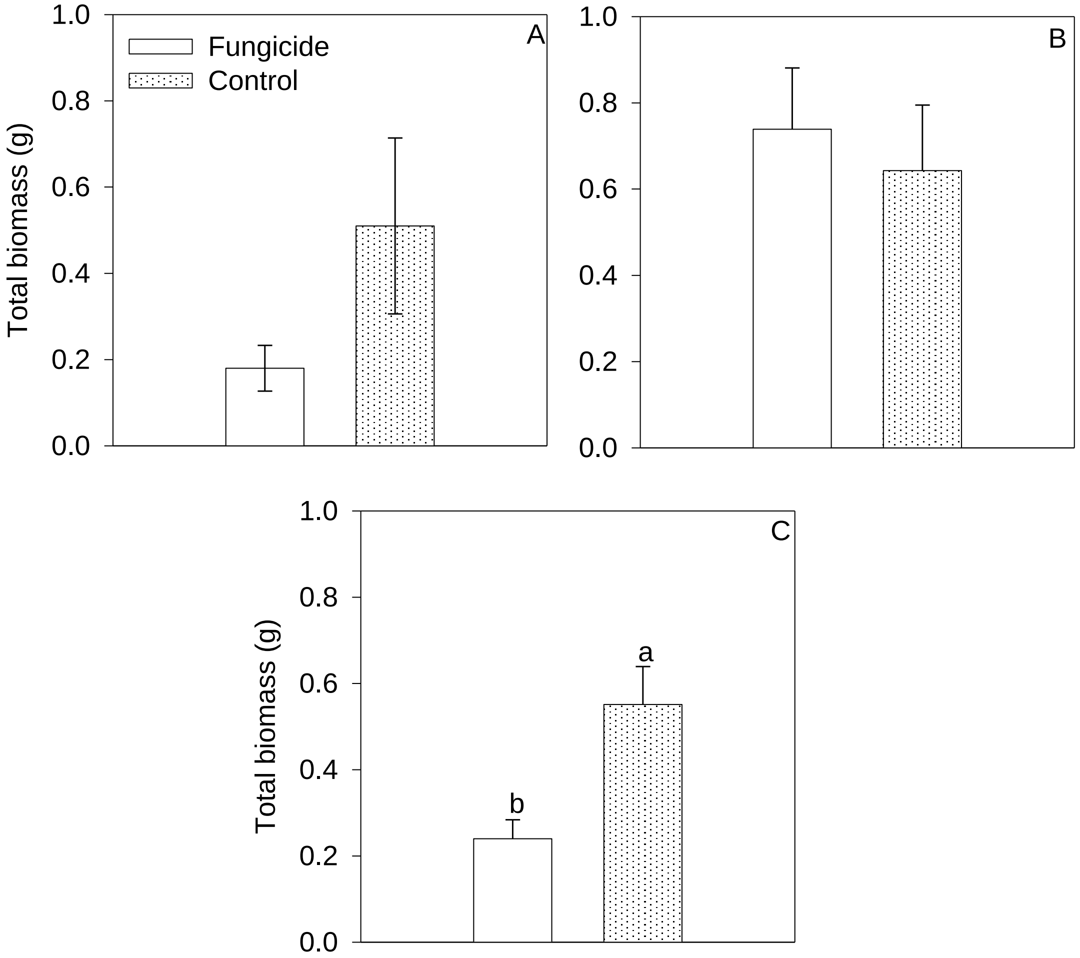

Figure 5.4. Experiment 1: The effect of (A) fungicide treatment on total biomass (±SE) of (A) *B. halimifolia*, (B) *M. cerifera*, and (C) *T. sebifera*. Different letters indicate significantly different means between sources (α = 0.05).

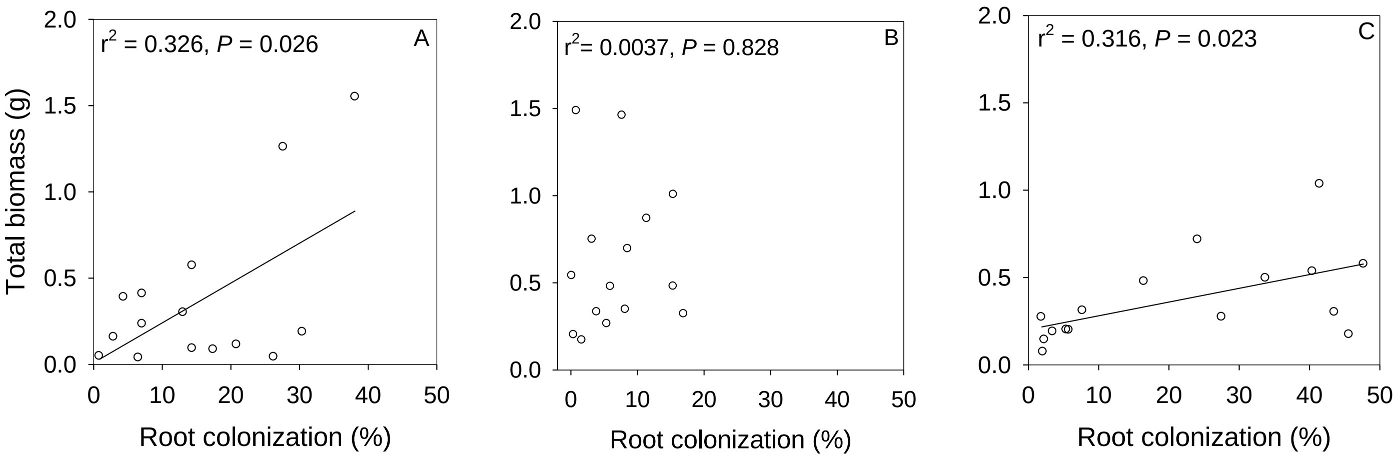

Figure 5.5. Experiment 1: The relationship between percent VAM colonization and total biomass of (A) *B. halimifolia*, (B) *M. cerifera*, and (C) *T. sebifera* seedlings grown in the greenhouse in fungicide treated and untreated field soils.

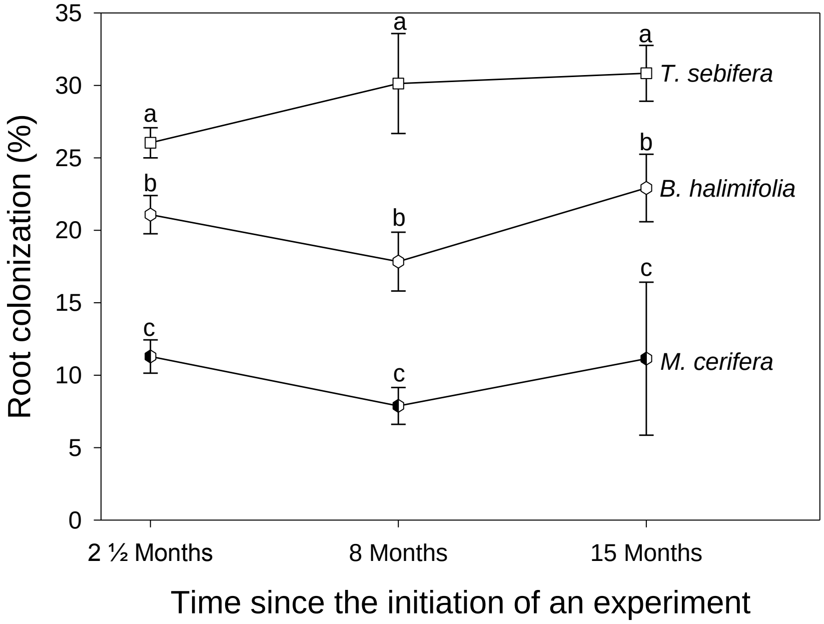

Figure 5.6. Experiment 2: The percentage VAM colonization (±SE) among species grown in the field after 2 ½ months (Sept 2010), 8 months (March 2011), and 15 months (Sept 2011) of growth. Comparisons were made among species only within each harvest and different letters indicate significantly different means (α = 0.05).

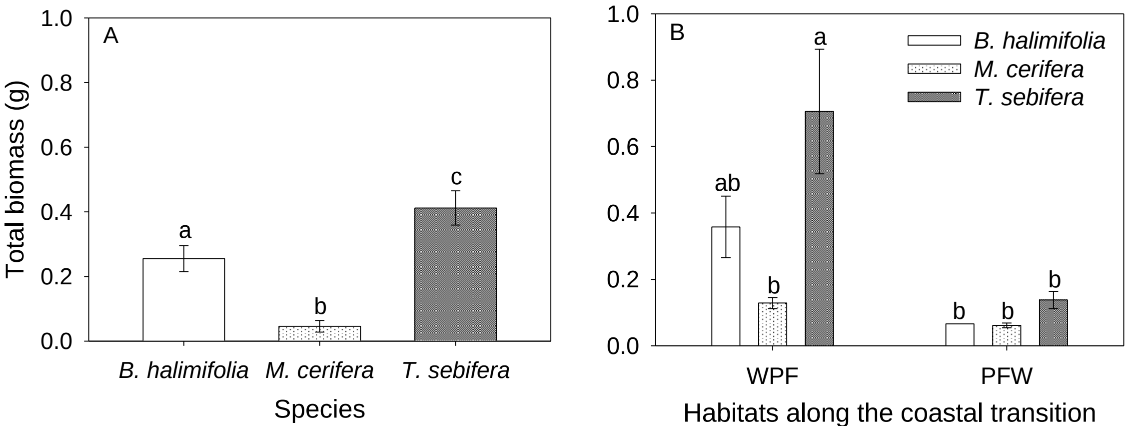

Figure 5.7. Experiment 2: (A) Variation in total dry biomass (Means ±SE) among species after 2½ months (Sept 2010) of growth, and (B) effect of habitats on total biomass among species after 8 months (March 2011) of growth. Different letters indicate significantly different means after pairwise comparisons (α = 0.05).

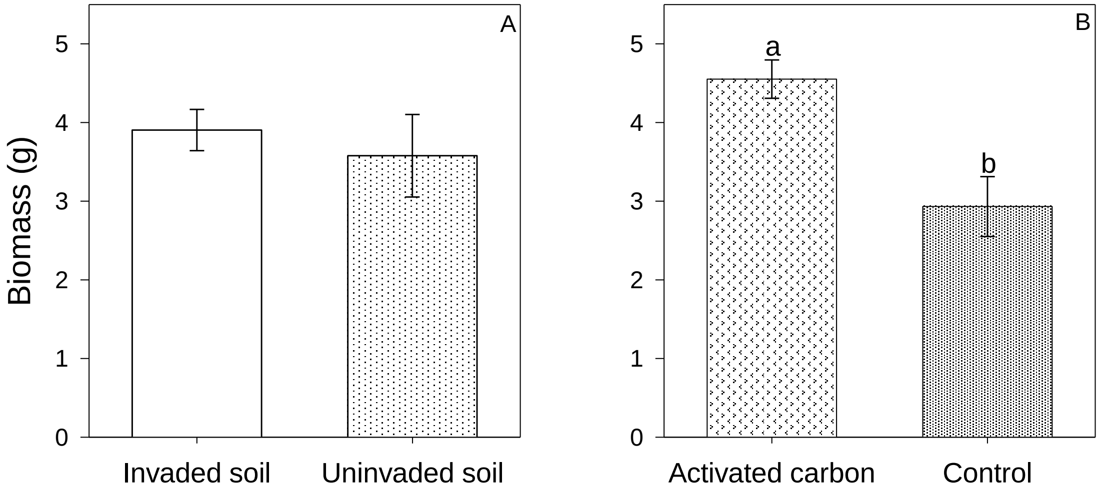

Figure 5.8. Experiment 3: Effects of (A) soil source (*T. sebifera* invaded or uninvaded) and (B) carbon source (activated carbon or control) on total dry biomass (±SE) of *B. halimifolia* seedlings. Different letters indicate significantly different means (α = 0.05).

CHAPTER 6

CONCLUSIONS

Low-lying coastal transition ecosystems at the interface of the land and sea are facing imminent threats from two components of global climate change, namely accelerated sea level rise and increased intensity of tropical storms. Disturbances produced by intense hurricanes are likely to have major effects on structure and stability of coastal ecosystems. A variety of recent publications revealed potential responses of plant species (both native and invasive) to various components of global change. The majority of those publications described the responses of native and invasive species to elevated temperature, CO_2, and precipitation (Smith et al. 2000, Ziska 2003, Walther 2010, Dukes et al. 2011). Despite the large body of recent research, both at the experimental and theoretical levels, it remains largely untested how invasive and co-occurring native species will respond to the future environmental conditions predicted with climate change, such as elevated salinity from sea level rise, and canopy damage and storm surge from hurricane winds. Species responses to environmental changes are important for formulating predictions about the future structure and dynamics of coastal plant communities.

In this dissertation, I first used a field observational study to assess the probability of occurrence of invasive *T. sebifera* and three co-occurring native species, *B. halimifolia*, *I. vomitoria*, and *M. cerifera* along a typical coastal transition of coastal Mississippi, USA. I related these species' occurrences to surrounding environmental variables (soil salinity, canopy openness, and other soil characteristics). To understand the responses of invasive and native species to environmental changes triggered by accelerated sea level rise and increased intensity of hurricanes, I used manipulative experiments and measured the species performances under controlled and simulated future environmental conditions. Additionally, I looked at the potential positive effects from soil biota (mycorrhizae) on *T. sebifera* and selected native species *B. halimifolia* and *M. cerifera* in producing positive responses to environmental changes. Below, I summarize the findings from my four primary data chapters.

Chapter II. Abiotic constraints on juvenile occurrence of invasive and native plant species: Implications for future distributions with climate. Understanding species' distribution in the field, by relating their presence and absence to surrounding environmental predictors, is fundamental for conservation planning (Funk and Vitousek 2007), and forecasting future distribution patterns under changing environmental conditions (Araujo and Williams 2000). Additionally, the invasive potential of exotic species can be predicted by knowing the invasive species' environmental breadth (Vazquez 2006). In this section, I selected a typical coastal transition at Grand Bay National Estuarine Research Reserve (GBNERR), coastal Mississippi, as a study system to look at the field distribution of invasive *T. sebifera* and co-occurring native species, *B. halimifolia*, *I. vomitoria*, and *M. cerifera* in relation to surrounding environmental predictors.

In an observational study all species showed individualistic relationships to environmental variables. The probability of occurrence of invasive *T. sebifera* and native *I. vomitoria*, and *M. cerifera* was constrained by soil water conductivity (i.e., salinity) and their presence was limited towards the seaward section of the coastal transition. In contrast, native *B. halimifolia* distribution was not affected much by soil water conductivity. These results suggest that increased salinity may have negative impacts on the distribution of the less salt tolerant invasive *T. sebifera* and native *I. vomitoria* and *M. cerifera*. These species may suffer a range contraction away from the coast in response to increased inundation triggered by sea level rise and storm surges. The invasive *T. sebifera* probability of occurrence was highest in areas that are in close proximity to anthropogenic activities (roads, power lines, rail road, water bodies, and recreational sites). It is predicted that the interactions between natural and anthropogenic disturbances may perhaps increase the dominance of invasive species including *T. sebifera*, by increasing resources and widening empty spaces, in more inland forested areas of the southeastern USA.

Chapter III. Germination response of the invasive* Triadica sebifera *and two co-occurring native woody species to elevated salinity across a Gulf Coast transition ecosystem. In the face of predicted increases in sea level rise and tidal inundation, low-lying coastal wetlands, such as those along the northern Gulf of Mexico, will experience severe risk of retreat (Williams et al. 1999a, Scavia et al. 2002, Battaglia et al. 2012). In particular, recruitment and establishment of many woody species of the forested wetlands will be impacted more severely compared to non-woody species (Kozlowski 1997). Thus, survival and persistence of coastal wetland species depend on their ability to tolerate salinity stress and increase their recruitment or migrate inland to keep pace with sea level rise. In this chapter, using growth chamber and greenhouse experiments, I tested germination responses of invasive *T. sebifera* and native *B. halimifolia* and *M. cerifera* to elevated salinity that is expected with sea level rise. Additionally, in the greenhouse, I also tested the germination of these species across a range of soil types collected from five dominant vegetation zones that define a typical coastal transition at GBNERR, coastal Mississippi, US. Successful germination may provide a basis for subsequent establishment and growth in a species (Donohue et al. 2010). I predict that germinable capability may elucidate the species’ persistence under changed environmental conditions.

The percent germination of all species was negatively impacted by elevated salinity in both the experiments. In the greenhouse, germination of all species was reduced in soils from the most seaward locations. Here, I also found species specific responses to salinity treatments. Among the three species, *B. halimifolia* was affected the least from elevated salinity. Invasive *T. sebifera* maintained nearly 10 % germination in inland soils even with highest level of salinity treatment (30 g/l), equivalent to full strength sea water. *M. cerifera* germination was affected the most with elevated salinity. These results suggest that the saltwater intrusion and resultant elevated salinity may not limit *T. sebifera*'s recruitment in inland forested areas. I predict that variation in germination responses to elevated salinity across coastal transition communities is likely to generate compositional shifts that include landward expansion of invasive *T. sebifera* throughout the forests of southeastern US.

Chapter IV. Responses of two native species and the invasive* Triadica sebifera *to simulated hurricane disturbances in forested ecosystems of coastal Mississippi, southeastern US. Hurricanes are major disturbances along the low-lying coastal areas of the Gulf Coast region that can affect plant communities directly through tree canopy damage, storm surges, soil erosion, debris and wrack deposition (Michener et al. 1997). However, hurricane wind disturbances and storm surge effects are highly variable and complex, depending on wind velocity, direction, site exposure, local topography, hydrology, soil types, stand structure, and site management history (Boose et al. 1994, Kupfer et al. 2008). Furthermore anthropogenic activities (e.g., management or disturbances) may also influence the effects of hurricanes (Grove et al. 2000). Despite the large body of work that describe hurricanes impacts on coastal forests (e.g., defoliation, canopy damage, uprooting trees, and mortality), biogeochemical cycles (Gresham et al. 1991, Hook et al. 1991, Gardner et al. 1992, Hopkinson et al. 2008, Busby et al.

2009), little work has been done to investigate the response of invasive and co-occurring native species, growing in different forest stands along coastal transitions, to increased hurricane wind damage and storm surges.

In this chapter, using a controlled greenhouse experiment, I tested seedling growth and mortality responses of invasive *T. sebifera* and two natives, *B. halimifolia* and *M. cerifera* to simulated hurricane disturbances (canopy openness and storm surge). In the experiment, I subjected seedlings to simulated pre-and post-hurricane canopy conditions, applied storm surges, and monitored seedlings growth and mortality for sixteenth months, covering two growing seasons before seedlings were harvested for biomass estimation. Although all the species benefitted from simulated post hurricane canopy openings, each species was affected differently in the two types of forest stands by simulated hurricane disturbances. Simulated storm surge effects were ephemeral and species recovered, mainly in open and/or high light treatments. However, storm surge produced greater seedling mortality under pre-hurricane canopy treatment in the simulated forest stands where natural disturbances have been suppressed. Among all the species, the invasive *T. sebifera* was the most shade tolerant compared to native species. These results implied that the high shade tolerance of *T. sebifera* likely provides this species a competitive edge over native species with open canopy conditions. Results from this study support the hypothesis that responses of native and invasive species to hurricane disturbances would differ at different forest stands along the coastal transitions. More importantly, due to high shade tolerance, long term dominance of *T. sebifera* can be expected in many coastal forested areas that are prone to hurricane disturbances.

Chapter V. The role of vesicular arbuscular mycorrhizae (VAM) on success of* Triadica sebifera *invasion in coastal transition ecosystems. Recent studies suggest that mutualistic associations between mycorrhizae fungi and plants play important roles in plant community structure and ecosystem functioning (Bever 1994, Klironomos 2002, Reynolds et al. 2003, Wardle et al. 2004). Beneficial root-colonizing mycorrhizal fungi may stimulate plant growth and development by enhancing plant nutrient uptake (e.g., phosphorus). It is well recognized that mutualistic mycorrhizal fungi such as vesicular arbuscular mycorrhizae (VAM) may stimulate plant resistance and/or tolerance to abiotic stresses (Auge 2001, Compant et al. 2010). Studies have also shown that VAM fungi can help plants to overcome salinity stress by increasing nutrient uptake and cation balance (Giri et al. 2007, Daei et al. 2009, Evelin et al. 2009). These positive effects may be significant for plant species inhabiting low-lying coastal wetlands that are predicted to experience more stressful saline conditions from unabated sea level rise and hurricane generated storm surges.

Recent studies have shown that some invasive plant species receive unusually strong positive effects from VAM compared to native species (Reinhart and Callaway 2006, Nijjer et al. 2008). Some invasive species are able to develop beneficial mutualisms with VAM at a higher rate than co-occurring native species (Marler et al. 1999b, Nijjer et al. 2008, Smith et al. 2008). I predicted that the the invasive *T. sebifera* would have a higher degree of VAM colonization, greater positive feedback from VAM, and greater salinity tolerance compared to native species. In this chapter, using a series of greenhouse and field experiments, I determined the degree of VAM colonization among the target species: *B. halimifolia*, *M. cerifera*, and *T. sebifera* then assessed the mycorrhizal dependency of these species for survival and growth. In addition, I tested the capacity of each species to tolerate salinity with the help of mutualistic VAM. Finally, I tested the possible allelopathic effects of the invasive *T. sebifera* on native species *B. halimifolia* and *M. cerifera*.

My results support the hypothesis that the invasive species would enjoy a higher degree of VAM colonization than native species. I found a significantly higher VAM colonization in *T. sebifera* than native species under both greenhouse and field conditions. Furthermore, *T. sebifera* showed higher VAM dependency on growth in the greenhouse experiment (significantly higher biomass in active soil), suggesting that *T. sebifera* may require a higher degree of mutualism with VAM for its initial establishment in the coastal ecosystems of the southeastern US. However, my study does not show positive effects of a higher degree of VAM colonization on salinity tolerance and survival of native and invasive. In fact, this potential effect could not be tested because simulated storm surge yielded complete mortality of *T. sebifera* as well as native *M. sebifera* seedlings, irrespective of antifungal treatment or not. However, *B. halimifolia* was able to recover from the surge, probably because of its inherently high salinity tolerance. These results however, do not undermine the utility of mycorrhizae in salinity tolerance of several other plants species, including many crop plants (Mukerji and Kapoor 1986, Feng et al. 2002, Giri et al. 2007). Results from the allelopathic experiment suggests that *T. sebifera* is not allelopathic in this coastal system and does not interfere with mutualistic relationships between VAM and native plant species, as suggested for other invasive species elsewhere (Stinson et al. 2006). Although, I did not find evidence of improved salinity tolerance for mycorrhizal seedlings, strong positive effects from VAM to *T. sebifera* may enhance the competitive capability of this species to invade coastal forests of the southeastern US.

While we debate about whether and how climate change benefits invasive species and which species are better suited to changing environmental conditions, some native species that are already at risk are being stressed by rapid sea level rise and storm surge in low-lying coastal areas. Findings of this research suggest that it is not just the characteristics of the native and invasive species that determine the trajectories of coastal plant communities, but also those of the communities impacted by tropical storms and inundation regimes. While low-lying coastal natural plant communities of the future may not have current day analogs, our job is explore the multifaceted effects of shifting disturbance regimes on invasive and co-occurring native species that would increase our ability predict habitat vulnerability to invasion. This is important in establishing science based conservation priorities in the critical marine-terrestrial ecotone areas that are threatened by sea level rise, shifting tropical storms regimes, and spread of invasive species.

Implications for Management and Species Responses to Environmental Changes -

Low-lying coastal transition ecosystems, such as the coastal ecosystems of the southeastern USA, will face direct impacts of climate change, particularly from a rising sea levels coupled with predicted increase in severity of intense tropical storms. These extreme natural events coupled with anthropogenic disturbances can create novel environmental conditions which are readily colonized by new exotic invasive species (Diez et al. 2012). Recent studies reported that the change in environmental condition due to sea level rise increased the prevalence of salt tolerant invasive *Phragmites australis* along the Atlantic coast, USA (Vasquez et al. 2005). Some other studies found that the intense tropical storms that increase resource opportunities through forest canopy disturbance may increase the prevalence of invasions of exotic species in the landscape (Bellingham et al. 2005, Chapman et al. 2008, Murphy et al. 2008b).

This study represents one approach of translating scientific findings into management recommendation for controlling exotic invasive species, which can possibly be replicated in other areas with similar geographical and environmental settings. The results presented here suggest that exotic *T. sebifera* invasion is associated with the presence of natural (e.g., canopy damage and soil erosion) and anthropogenic disturbances (e.g., active roads, trails, abandoned settlements, power line, and natural fire suppression, etc.). These results provide important information about the target areas (e.g., close to active roads, water bodies, abandonment settlements, and increasing presence of anthropogenic activities) where management and mitigation of *T. sebifera* invasion should be targeted. A simple inventory approach, used here (Chapter 2), is useful for identifying areas at risk of *T. sebifera* invasion, prioritizing control efforts, and investigating the role of surrounding environmental conditions on future invasion probabilities. Additionally, increased probability of occurrence of young juveniles of *T. sebifera* in fire suppressed forest stands of dense tree canopy of its own imply that the low light availability reduce the chances of regeneration of less shade tolerant native tree seedlings. While *T. sebifera* is increasingly more prevalent in coastal Mississippi, early detection of sites colonized by this invasive species would aid in rapid management decisions that may abate the invasion process and enhance the resiliency and adaptability of natural communities to predicted environmental changes.

The experimental studies that manipulated the future environmental conditions and examined the specific impacts of these conditions on performances of invasive *T. sebifera* and co-occurring native species have provided important implications for forecasting responses of native and invasive species to effects of climate change (e.g., sea level rise and tropical storms). These studies also help in identifying the potential for future spread of *T. sebifera* in low lying coastal ecosystems. Results from these experimental studies suggest that the responses of native and invasive species to effects of climate change and altered environmental conditions are not straightforward. Native and invasive species my take separate response trajectories and invasive *T. sebifera* have the potential to perform better under some environmental conditions due to its ability to cope with broad range of environmental stress (Chapter 2,3, and 4). A complex interaction between natural and anthropogenic disturbances, habitat types and their position in the landscape, stand structure, soil biota, and the legacy of land use patterns may determine the responses of native and invasive species to future environmental changes. These interactions may constitute the structure and functions of the coastal ecosystems. Thus, a coordinated multidisciplinary research activities are needed to enhance our predictive capabilities for future environmental changes and identify the links of natural and anthropogenic disturbances that may exacerbate the invasion process at local and regional scale. The ability to understand severity of environmental change and species responses to these changes would ultimately guide the management options for limiting invasions.

REFERENCES

Abbaspour, H., S. Saeidi-Sar, H. Afshari, and M. A. Abdel-Wahhab. 2012. Tolerance of Mycorrhiza infected Pistachio (*Pistacia vera* L.) seedling to drought stress under glasshouse conditions. Journal of Plant Physiology **169**:704-709.

Agrawal, A. A., P. M. Kotanen, C. E. Mitchell, A. G. Power, W. Godsoe, and J. Klironomos. 2005. Enemy release? An experiment with congeneric plant pairs and diverse above- and belowground enemies. Ecology **86**:2979-2989.

Allen, C. D., A. K. Macalady, H. Chenchouni, D. Bachelet, N. McDowell, M. Vennetier, T. Kitzberger, A. Rigling, D. D. Breshears, E. H. Hogg, P. Gonzalez, R. Fensham, Z. Zhang, J. Castro, N. Demidova, J.-H. Lim, G. Allard, S. W. Running, A. Semerci, and N. Cobb. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management **259**:660-684.

Allen, E. B. and M. E. Allen. 1990. The mediation of competition by mycorrhizae in successional and patchy environments. Pages 367-389 *in* J. B. Grace and D. Tilman, editors. Perspectives on plant competition. Academic Press, New York, USA.

Allison, P. D. 1999. Logistic regression using SAS system: theory and application. SAS Institute Inc., Cary, NC, USA.

Alpert, P., E. Bone and C. Holzapfel (2000) Invasiveness, invasibility and the role of environmental stress in the spread of non-native plants. Perspective in Plant Ecolology Evolution and Systematic **3:** 52-66.

Alston, K. P. and D. M. Richardson. 2006. The roles of habitat features, disturbance, and distance from putative source populations in structuring alien plant invasions at the urban/wildland interface on the Cape Peninsula, South Africa. Biological Conservation **132**:183-198.

Anderegg, W. R. L., J. M. Kane, and L. D. L. Anderegg. 2012. Consequences of widespread tree mortality triggered by drought and temperature stress. Nature Climate Change. doi:10.1038/nclimate1635.

Araujo, M. B. and P. H. Williams. 2000. Selecting areas for species persistence using occurrence data. Biological Conservation **96**:331-345.

Auge, R. M. 2001. Water relations, drought and vesicular-arbuscular mycorrhizal symbiosis. Mycorrhiza **11**:3-42.

Baker, H. 1974. The evolution of weeds. Annual Review of Ecology and Systematics **5**:1–24.

Bandou, E., F. Lebailly, F. Muller, M. Dulormne, A. Toribio, J. Chabrol, R. Courtecuisse, C. Plenchette, Y. Prin, R. Duponnois, M. Thiao, S. Sylla, B. Dreyfus, and A. M. Ba. 2006. The ectomycorrhizal fungus Scleroderma bermudense alleviates salt stress in seagrape (Coccoloba uvifera L.) seedlings. Mycorrhiza **16**:559-565.

Barea, J. M., R. Azcon, and C. Azcon-Aguilar. 2002. Mycorrhizosphere interactions to improve plant fitness and soil quality. International Journal of General and Molecular Microbiology **81**:343-351.

Barrilleaux, T. C. and J. B. Grace. 2000. Growth and invasive potential of Sapium sebiferum (Euphorbiaceae) within the coastal prairie region: The effects of soil and moisture regime. American Journal of Botany **87**:1099-1106.

Baskin, C. C. and J. M. Baskin. 1998. Seeds: Ecology, biogeography, and evolution of dormancy and germination. Academic Press, San Diego, CA.

Batista, W. B. and W. J. Platt. 2003. Tree population responses to hurricane disturbance: syndromes in a south-eastern USA old-growth forest. Journal of Ecology **91**:197-212.

Battaglia, L. L., J. S. Denslow, J. R. Inczauskis, and S. G. Baer. 2009. Effects of native vegetation on invasion success of Chinese tallow in a floating marsh ecosystem. Journal of Ecology **97**:239-246.

Battaglia, L. L., R. R. Sharitz, and P. R. Minchin. 1999. Patterns of seedling and overstory composition along a gradient of hurricane disturbance in an old-growth bottomland hardwood community. Canadian Journal of Forest Research **29**:144-156.

Battaglia, L. L., M. S. Woodrey, M. S. Peterson, K. S. Dillon, and J. M. Visser. 2012. Wetlands of the Northern Gulf Coast. Pages 75-88 *in* D. P. Batzer and A. H. Baldwin, editors. Wetland habitats of North America, University of California Press, Berkeley.

Beckage, B., J. S. Clark, B. D. Clinton, and B. L. Haines. 2000. A long-term study of tree seedling recruitment in southern Appalachian forests: the effects of canopy gaps and shrub understories. Canadian Journal of Forest Research **30**:1617-1631.

Bellingham, P. J., E. V. J. Tanner, and J. R. Healey. 1995. Damage and responsiveness to Jamaican Montane tree species after disturbance by a hurricane. Ecology **76**:2562-2580.

Bellingham, P. J., E. V. J. Tanner, and J. R. Healey. 2005. Hurricane disturbance accelerates invasion by the alien tree *Pittosporum undulatum* in Jamaican montane rain forests. Journal of Vegetation Science **16**:675-684.

Bellingham, P. J., E. V. J. Tanner, P. M. Rich, and T. C. R. Goodland. 1996. Changes in light below the canopy of a Jamaican montane rainforest after a hurricane. Journal of Tropical Ecology **12**:699-722.

Bender, M. A., T. R. Knutson, R. E. Tuleya, J. J. Sirutis, G. A. Vecchi, S. T. Garner, and I. M. Held. 2010. Modeled impact of anthropogenic warming on the frequency of intense Atlantic hurricanes. Science **327**:454-458.

Bernstein, L. 1975. Effects of salinity and sodicity on plant growth. Annual Review of Phytopathology **13**: 295-312.

Bever, J. D. 1994. Feedback between plants and their soil communities in an old field community. Ecology **75**:1965-1977.

Bever, J. D., K. M. Westover, and J. Antonovics. 1997. Incorporating the soil community into plant population dynamics: the utility of the feedback approach. Journal of Ecology **85**:561-573.

Birken, A. S. and D. J. Cooper. 2006. Processes of Tamarix invasion and floodplain development along the lower Green River, Utah. Ecological Applications **16**:1103-1120.

Blumenthal, D. M. 2006. Interactions between resource availability and enemy release in plant invasion. Ecology Letters **9**:887-895.

Boose, E. R., D. R. Foster, and M. Fluet. 1994. Hurricane impacts to tropical and temperate forest landscapes. Ecological Monographs **64**:369-400.

Bower, M. J., C. E. Aslan, and M. and Rejmánek. 2009. Invasive potential of Chinese tallow (Triadica sebifera) in California's Central Valley. Invasive Plant Science and Management **2**:386-395.

Bradley, B. A., D. M. Blumenthal, R. Early, E. D. Grosholz, J. J. Lawler, L. P. Miller, C. J. B. Sorte, C. M. D'Antonio, J. M. Diez, J. S. Dukes, I. Ibanez, and J. D. Olden. 2012. Global change, global trade, and the next wave of plant invasions. Frontiers in Ecology and the Environment **10**:20-28.

Bradley, B. A., D. M. Blumenthal, D. S. Wilcove, and L. H. Ziska. 2010. Predicting plant invasions in an era of global change. Trends in Ecology & Evolution **25**:310-318.

Bradley, B. A., M. Oppenheimer, and D. S. Wilcove. 2009. Climate change and plant invasions: restoration opportunities ahead? Global Change Biology **15**:1511-1521.

Bradley, B. A. and D. S. Wilcove. 2009. When invasive plants disappear: transformative restoration possibilities in the western United States resulting from climate change. Restoration Ecology **17**:715-721.

Bradley, B. A. and J. F. Mustard. 2006. Characterizing the landscape dynamics of an invasive plant and risk of invasion using remote sensing. Ecological Application **16**:1132-1147.

Brinkman, E. P., W. H. Van der Putten, E.-J. Bakker, and K. J. F. Verhoeven. 2010. Plant-soil feedback: experimental approaches, statistical analyses and ecological interpretations. Journal of Ecology **98**:1063-1073.

Brown D. R., T. W. Sherry, and J. Harris. 2011. Hurricane Katrina impacts the breeding bird community in a bottomland hardwood forest of the Pearl River basin, Louisiana. Forest Ecology and Management **261**:111-119.

Brown, J. H., T. J. Valone, and C. G. Curtin. 1997. Reorganization of an arid ecosystem in response to recent climate change. Proceedings of the National Academy of Sciences of the United States of America **94**:9729-9733

Bruce, K. A. 1993. Factors affecting the biological invasion of the exotic Chinese tallow tree, *Sapium sebiferum*, in the Gulf coast prairie of Texas. Masters Thesis, University of Houston, Houston, Texas, USA.

Bruce, K. A., G. N. Cameron, and P. A. Harcombe. 1995. Initiation of a New Woodland Type on the Texas Coastal Prairie by the Chinese Tallow Tree (*Sapium sebiferum* (L.) Roxb.). Bulletin of the Torrey Botanical Club **122**:215-225.

Bruce, K. A., G. N. Cameron, P. A. Harcombe, and G. Jubinsky. 1997. Introduction, impact on native habitats, and management of a woody invader, the Chinese tallow tree, *Sapium sebiferum* (L) Roxb. Natural Areas Journal **17**:255-260.

Burgiel, S. W. and A. A. Muir. 2010. Invasive species, climate change and ecosystem based adaptation: addressing multiple drivers of global change. Global Invasive Species Programme (GISP), Washington, DC, US, and Nairobi, Kenya.

Burns, J. H., T. E. Miller, and Fw. 2004. Invasion of Chinese tallow (*Sapium sebiferum*) in the Lake Jackson area, northern Florida. American Midland Naturalist **152**:410-417.

Busby, P. E., C. D. Canham, G. Motzkin, and D. R. Foster. 2009. Forest response to chronic hurricane disturbance in coastal New England. Journal of Vegetation Science **20**:487-497.

Busby, P. E., G. Motzkin, and E. R. Boose. 2008. Landscape-level variation in forest response to hurricane disturbance across a storm track. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere **38**:2942-2950.

Callaway, R. M. and E. T. Aschehoug. 2000. Invasive plants versus their new and old neighbors: A mechanism for exotic invasion. Science **290**:521-523.

Callaway, R. M., E. J. Bedmar, K. O. Reinhart, C. Gomez Silvan, and J. Klironomos. 2011. Effects of soil biota from different ranges on *Robinia* invasion: acquiring mutualists and escaping pathogens. Ecology **92**:1027-1035.

Callaway, R. M., B. Newingham, C. A. Zabinski, and B. E. Mahall. 2001. Compensatory growth and competitive ability of an invasive weed are enhanced by soil fungi and native neighbours. Ecology Letters **4**:429-433.

Callaway, R. M. and W. M. Ridenour. 2004. Novel weapons: invasive success and the evolution of increased competitive ability. Frontiers in Ecology and the Environment **2**:436-443.

Callaway, R. M., G. C. Thelen, A. Rodriguez, and W. E. Holben. 2004. Soil biota and exotic plant invasion. Nature **427**:731-733.

Canham, C. D. and P. L. Marks. 1985. The response of woodyplants to disturbance: patterns of establishment and growth. Pages 197-216 *in* S. T. A. Pickett and P. White, editors. The ecology of natural disturbance and patch dynamics. Academic Press, Orlando.

Chambers, J. Q., J. I. Fisher, H. C. Zeng, E. L. Chapman, D. B. Baker, and G. C. Hurtt. 2007. Hurricane Katrina's carbon footprint on U. S. Gulf Coast forests. Science **318**:1107-1107.

Chambless, L. F. and E. S. Nixon. 1975. Woody vegetation-soil relations in a bottomland forest of east Texas. Texas Journal of Science **26**:407-416.

Chapman, E. L., J. Q. Chambers, K. F. Ribbeck, D. B. Baker, M. A. Tobler, H. C. Zeng, and D. A. White. 2008. Hurricane Katrina impacts on forest trees of Louisiana's Pearl River basin. Forest Ecology and Management **256**:883-889.

Cleland, E. E., J. M. Allen, T. M. Crimmins, J. A. Dunne, S. Pau, S. E. Travers, E. S. Zavaleta, and E. M. Wolkovich. 2012. Phenological tracking enables positive species responses to climate change. Ecology **93**:1765-1771.

Cleverly, J. R., S. D. Smith, A. Sala, and D. A. Devitt. 1997. Invasive capacity of *Tamarix ramosissima* in a Mojave Desert floodplain: The role of drought. Oecologia **111**:12-18.

Cohen, A. N. and J. T. Carlton. 1998. Accelerating invasion rate in a highly invaded estuary. Science **279**: 555-558.

Compant, S., M. G. A. van der Heijden, and A. Sessitsch. 2010. Climate change effects on beneficial plant-microorganism interactions. Fems Microbiology Ecology **73**:197-214.

Conner, W. H. 1994. The effect of salinity and waterlogging on growth and survival of baldcypress and Chinese tallow seedlings. Journal of Coastal Research **10**:1045-1049.

Conner, W. H. and G. R. Askew. 1993. Impact of saltwater flooding on Red Maple, Redbay, and Chinese tallow seedlings. Castanea **58**:214-219.

Conner, W. H., K. W. McLeod, and J. K. McCarron. 1997. Flooding and salinity effects on growth and survival of four common forested wetland species. Wetlands Ecology and Management **5**:99-109.

Conner, W. H., L. Mihalia, and J. Wolfe. 2002. Tree community structure and changes from 1987 to 1999 in three Louisiana and three South Carolina forested wetlands. Wetlands **22**:58-70.

Conner, W. H., W. D. Mixon, and G. W. Wood. 2005. Maritime forest habitat dynamics on Bulls Island, Cape Romain National Wildlife Refuge, SC, following hurricane Hugo. Forest Ecology and Management **212**:127-134.

Conway, W. C., L. M. Smith, and J. F. Bergan. 2002. Potential allelopathic interference by the exotic Chinese tallow tree (Sapium sebiferum). American Midland Naturalist **148**:43-53.

Cooper-Ellis, S., D. R. Foster, G. Carlton, and A. Lezberg. 1999. Forest response to catastrophic wind: Results from an experimental hurricane. Ecology **80**:2683-2696.

Crain, C. M., B. R. Silliman, S. L. Bertness, and M. D. Bertness. 2004. Physical and biotic drivers of plant distribution across estuarine salinity gradients. Ecology **85**:2539-2549.

Debez, A., H. Koyro, C. Grignon, C. Abdelly, and B. Huchzermeyer. 2008. Relationship between the photosynthetic activity and performance of *Cakile maritima* after long-term salt treatment. *Physiologia Plantarum* **133**: 373-385.

Daehler, C. C. 2003. Performance comparisons of co-occurring native and alien invasive plants: implications for conservation and restoration. Annual Review of Ecology Evolution and Systematics **34**:183-211.

Daei, G., M. R. Ardekani, F. Rejali, S. Teimuri, and M. Miransari. 2009. Alleviation of salinity stress on wheat yield, yield components, and nutrient uptake using arbuscular mycorrhizal fungi under field conditions. Journal of Plant Physiology **166**:617-625.

Dale, V. H., L. A. Joyce, S. McNulty, and R. P. Neilson. 2000. The interplay between climate change, forests, and disturbances. Science of the Total Environment **262**:201-204.

Dale, V. H., L. A. Joyce, S. McNulty, R. P. Neilson, M. P. Ayres, M. D. Flannigan, P. J. Hanson, L. C. Irland, A. E. Lugo, C. J. Peterson, D. Simberloff, F. J. Swanson, B. J. Stocks, and B. M. Wotton. 2001. Climate change and forest disturbances. Bioscience **51**:723-734.

Daughtrey, M. 2011. Cyclamen disease cycles. Pages 82-84 Grower Talks.

Davidson, A. M., M. Jennions, and A. B. Nicotra. 2011. Do invasive species show higher phenotypic plasticity than native species and, if so, is it adaptive? A meta-analysis. Ecology Letters **14**:419-431.

Davis, M. A., J. P. Grime, and K. Thompson. 2000. Fluctuating resources in plant communities: a general theory of invasibility. Journal of Ecology **88**:528-534.

Day, J. W., D. F. Boesch, E. J. Clairain, G. P. Kemp, S. B. Laska, W. J. Mitsch, K. Orth, H. Mashriqui, D. J. Reed, L. Shabman, C. A. Simenstad, B. J. Streever, R. R. Twilley, C. C.

Watson, J. T. Wells, and D. F. Whigham. 2007. Restoration of the Mississippi Delta: lessons from hurricanes Katrina and Rita. Science **315**:1679-1684.

Denslow, J. S. 1987. Tropical rainforest gaps and tree species diversity. Annual Review of Ecology and Systematics **18**:431-451.

Denslow, J. S. and L. L. Battaglia. 2002. Stand composition and structure across a changing hydrologic gradient: Jean Lafitte National Park, Louisiana, USA. Wetlands **22**:738-752.

Devall, M. S. and B. R. Parresol. 1998. Effects of global climate change on biodiversity in forests of the southern United States. Pages 663–668 *in* R. A. Mickler and S. Fox, editors. The productivity and sustainability of southern forest ecosystems in a changing environment. Springer-Verlag, New York, New York, USA.

DeWalt, S. J., J. S. Denslow, and K. Ickes. 2004. Natural-enemy release facilitates habitat expansion of the invasive tropical shrub *Clidemia hirta*. Ecology **85**:471-483.

Diez, J. M., C. M. D'Antonio, J. S. Dukes, E. D. Grosholz, J. D. Olden, C. J. B. Sorte, D. M. Blumenthal, B. A. Bradley, R. Early, I. Ibanez, S. J. Jones, J. J. Lawler, and L. P. Miller. 2012. Will extreme climatic events facilitate biological invasions? Frontiers in Ecology and the Environment **10**:249-257.

Donahue, C., W. E. Rogers, and E. Siemann. 2004. Effects of temperature and mulch depth on Chinese tallow tree (*Sapium sebiferum*) seed germination. Texas Journal of Science **56**:347-356.

Donnelly, J. P. and M. D. Bertness. 2001. Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated sea-level rise. Proceedings of the National Academy of Sciences of the United States of America **98**:14218-14223.

Donohue, K., R. R. de Casas, L. Burghardt, K. Kovach, and C. G. Willis. 2010. Germination, postgermination adaptation, and species ecological ranges. Annual Review of Ecology, Evolution, and Systematics **41**:293–319.

Drake, D. R. and I. A. Ungar. 1989. Effects of salinity, nitrogen, and population density on the survival, growth, and reproduction of *Atriplex triangularis* (Chenopodiaceae). American Journal of Botany **76***:* 1125-*1135.*

Dukes, J. S. 2002. Comparison of the effect of elevated CO2 on an invasive species (*Centaurea solstitialis* L.) in monoculture and community settings. Plant Ecology **160**:225-234.

Dukes, J. S., N. R. Chiariello, S. R. Loarie, and C. B. Field. 2011. Strong response of an invasive plant species (*Centaurea solstitialis* L.) to global environmental changes. Ecological Applications **21**:1887-1894.

Dukes, J. S. and H. A. Mooney. 1999. Does global change increase the success of biological invaders? Trends in Ecology & Evolution **14**:135-139.

Duncan, W. H. and M. B. Duncan. 1987. The Smithsonian guide to seaside plants of the Gulf and Atlantic coasts from Louisiana to Massachusetts, exclusive of lower peninsular Florida. Smithsonian Institution Press, Washington, DC.

Duncan, W. H., P. L. Piercy, S. D. Feurt, and R. Starling. 1957. Toxicological studies of southeastern plants. II. Compositae. Economic Botany **11**:75-85.

Ehrenfeld, J. G., P. Kourtev, and W. Z. Huang. 2001. Changes in soil functions following invasions of exotic understory plants in deciduous forests. Ecological Applications **11**:1287-1300.

Eleuterius, L. N. 1972. The Marshes of Mississippi. Castanea **37**:153-168.

Entry, J. A., P. T. Rygiewicz, L. S. Watrud, and P. K. Donnelly. 2002. Influence of adverse soil conditions on the formation and function of arbuscular mycorrhizas. Advances in Environmental Research **7**:123-138.

Ervin, G. N. 2009. Distribution, habitat characteristics, and new county-level records of *Baccharis halimifolia* L. on a portion of its present US range boundary. Southeastern Naturalist **8**:293-304.

Evans, E. D., Y. Anjaneyulu, and P. B. Tchnouwou. 2012. Effects of hurricane Katrina on land cover within the Grand Bay National Estuarine Research Reserve in Mississippi, USA, pp 173-188. *In* Vitale, K., (Ed.), Environmental and Food Safety and Security for South-East Europe and Ukraine, NATO Science for Peace and Security Series C: Environmental Security. Springer.

Evelin, H., R. Kapoor, and B. Giri. 2009. Arbuscular mycorrhizal fungi in alleviation of salt stress: a review. Annals of Botany **104**:1263-1280.

Ewel, J. J. 1986. Invasibility: lessons from south Florida. Pages 214-230 *in* H. A. Mooney and J. A. Drake, editors. Ecology of biological invasions of North America and Hawaii. Springer-Verlag, New York, USA.

Feng, G., F. S. Zhang, X. L. Li, C. Y. Tian, C. Tang, and Z. Rengel. 2002. Improved tolerance of maize plants to salt stress by arbuscular mycorrhiza is related to higher accumulation of soluble sugars in roots. Mycorrhiza **12**:185-190.

Flack, S. and E. Furlow. 1996. America’s least wanted "purple plague," "green cancer" and 10 other ruthless environmental thugs. Nature Conservancy Magazine.

Flory, S. L. and K. Clay. 2009. Effects of roads and forest successional age on experimental plant invasions. Biological Conservation **142**:2531-2537.

Flowers, T. J. and T. D. Colmer. 2008. Salinity tolerance in halophytes. New Phytologist **179**: 945-963.

Foster, D. R. 1988. Disturbance history, community organization and vegetation dynamics of the old-growth Pisgah forest, Southwestern New-Hampshire, USA. Journal of Ecology **76**:105-134.

Foster, D. R., D. H. Knight, and J. F. Franklin. 1998. Landscape patterns and legacies resulting from large, infrequent forest disturbances. Ecosystems **1**:497-510.

Frazer, G. W., C. D. Canham, and K. P. Lertzman. 1999. Gap Light Analyser (GLA). Version 2.0: Imaging software to extract canopy structure and gap light transmission indices from true-colour fisheye photographs. Users manual and program documentation, Simon Fraser University, Burnaby, B.C. and the Institute of Ecosystem Studies, Millbrook, New York.

Fritz, H. M., C. Blount, R. Sokoloski, J. Singleton, A. Fuggle, B. G. McAdoo, A. Moore, C. Grass, and B. Tate. 2007. Hurricane Katrina storm surge distribution and field observations on the Mississippi Barrier Islands. Estuarine Coastal and Shelf Science **74**:12-20.

Funk, J. L. and P. M. Vitousek. 2007. Resource-use efficiency and plant invasion in low-resource systems. Nature **446**:1079-1081.

Gan, J. B., J. H. Miller, H. H. Wang, and J. W. Taylor. 2009. Invasion of tallow tree into southern US forests: influencing factors and implications for mitigation. Canadian Journal of Forest Research **39**:1346-1356.

Gardner, L. R., W. K. Michener, T. M. Williams, E. R. Blood, B. Kjerve, L. A. Smock, D. J. Lipscomb, and C. Gresham. 1992. Disturbance effects of Hurricane Hugo on a pristine coastal landscape: North inlet, South Carolina, USA. Netherlands Journal of Sea Research **30**:249-263.

George, E., H. Marschner, and I. Jakobsen. 1995. Role of arbuscular mycorrhizal fungi in uptake of phosphorus and nitrogen from soil. Critical Reviews in Biotechnology **15**:257-270.

Ghalambor, C. K., J. K. McKay, S. P. Carroll, and D. N. Reznick. 2007. Adaptive versus non-adaptive phenotypic plasticity and the potential for contemporary adaptation in new environments. Functional Ecology **21**:394-407.

Gilliam, F. S. and W. J. Platt. 1999. Effects of long-term fire exclusion on tree species composition and stand structure in an old-growth *Pinus palustris* (Longleaf pine) forest. Plant Ecology **140**:15-26.

Giorgis, M. A., P. A. Tecco, A. M. Cingolani, D. Renison, P. Marcora, and V. Paiaro. 2011. Factors associated with woody alien species distribution in a newly invaded mountain system of central Argentina. Biological Invasions **13**:1423-1434.

Giri, B., R. Kapoor, and K. G. Mukerji. 2007. Improved tolerance of Acacia nilotica to salt stress by arbuscular mycorrhiza, *Glomus fasciculatum* may be partly related to elevated K/Na ratios in root and shoot tissues. Microbial Ecology **54**:753-760.

Godfrey, R. K. 1988. Trees, shrubs, and woody vines of northern Florida and adjacent Georgia and Alabama. University of Georgia Press, Athens, Georgia, USA.

Goldstein, D., D. R. Drake, C. Alpha, P. Melcher, J. Heraux, and A. Azocar. 1996. Growth and photosynthetic responses of *Scaevola sericea*, a Hawaiian coastal shrub, to substrate salinity and salt spray. International Journal of Plant Sciences **157***:* 171-*179.*

Goode, L. K. and M. F. Allen. 2009. Seed germination conditions and implications for establishment of an epiphyte, *Aechmea bracteata* (Bromeliaceae). Plant Ecology **204**:179-188.

Gorai, M. and M. Neffati. 2007. Germination responses of *Reaumuria vermiculata* to salinity and temperature. Annals of Applied Biology **151**:53-59.

Grabe, D. F. 1970. Tetrazolium testing handbook for agricultural seeds. Contribution No. 29 to the handbook on seed testing. Association of official seed analysts, Lincoln, NE.

Graves, W. R. and J. L. Gallagher. 2003. Resistance to salinity of *Alnus maritima* from disjunct wetlands: symptoms of salt injury, comparison to other shrubs, and effect of inundation. Wetlands **23**:394-405.

Greening, H., P. Doering, and C. Corbett. 2006. Hurricane impacts on coastal ecosystems. Estuaries and Coasts **29**:877-879.

Gresham, C. A. 1994. Tallow tree (*Sapium sebiferum* L.) allelopathy affects loblolly pine seed germination and seedling growth, p. 338-342. In: Edwards, M. Boyd (comp). Proc. 8th. biennial southern silvicultural research conference, general technical report SRS-1, Asheville, North Carolina: U.S. Department of Agriculture, Forest Service, Southern Research.

Gresham, C. A., T. M. Williams, and D. J. Lipscomb. 1991. Hurricane Hugo wind damage to southeastern United States coastal forest tree species. Biotropica **23**:420-426.

Grove, S. J., S. M. Turton, and D. T. Siegenthaler. 2000. Mosaics of canopy openness induced by tropical cyclones in lowland rain forests with contrasting management histories in northeastern Australia. Journal of Tropical Ecology **16**:883-894.

Grubb, P. J. 1977. Maintenance of species richness in plant communities - importance of regeneration niche. Biological Reviews of the Cambridge Philosophical Society **52**:107-145.

Hagerman, S. M., M. D. Jones, G. E. Bradfield, M. Gillespie, and D. M. Durall. 1999. Effects of clear-cut logging on the diversity and persistence of ectomycorrhizae at a subalpine forest. Canadian Journal of Forest Research **29**:124-134.

Harper, R. M. 1913. A botanical cross-section of northern Mississippi, with notes on the influence of soil on vegetation. Bulletin of the Torrey Botanical Club **40**:377-399.

Harper, J. L. 1977. Population biology of plants. Academic Press, New York, NY.

Hartnett, D. C. and G. W. T. Wilson. 1999. Mycorrhizae influence plant community structure and diversity in tallgrass prairie. Ecology **80**:1187-1195.

He, W.-M., J.-J. Li, and P.-H. Peng. 2012. A congeneric comparison shows that experimental warming enhances the growth of invasive *Eupatorium adenophorum*. Plos One **7**:e35681.

Hellmann, J. J., J. E. Byers, B. G. Bierwagen, and J. S. Dukes. 2008. Five potential consequences of climate change for invasive species. Conservation Biology **22**:534-543.

Hierro, J. L. and R. M. Callaway. 2003. Allelopathy and exotic plant invasion. Plant and Soil **256**:29-39.

Hierro, J. L., J. L. Maron, and R. M. Callaway. 2005. A biogeographical approach to plant invasions: the importance of studying exotics in their introduced and native range. Journal of Ecology **93**:5-15.

Hilbert, K. W. 2006. Land cover change within the Grand Bay National Estuarine Research Reserve: 1974-2001. Journal of Coastal Research **22**:1552-1557.

Hobbs, R. J. 1989. The nature and effects of disturbance relative to invasions. Pages 389-405 *in* H. A. M. J. A. Drake, F. di Castri, IL H. Groves, F.J. Kruger, M. Rejmanek, and M. and Williamson, editors. Biological invasions. A global perspective. Wiley, Chichester, England.

Hobbs, R. J. and L. F. Huenneke. 1992. Disturbance, diversity, and invasion - implications for conservations. Conservation Biology **6**:324-337.

Hodge, A. and A. H. Fitter. 2010. Substantial nitrogen acquisition by arbuscular mycorrhizal fungi from organic material has implications for N cycling. Proceedings of the National Academy of Sciences of the United States of America **107**:13754-13759.

Holway, D. A., A. V. Suarez, and T. J. Case. 2002. Role of abiotic factors in governing susceptibility to invasion: A test with argentine ants. Ecology **83**:1610-1619.

Honig, M. A., R. M. Cowling, and D. M. Richardson. 1992. The invasive potential of Australian Banksias in South-African fynbos - A comparison of the reproductive potential of *Banksia ericifolia* and *Leucadendron laureolum*. Australian Journal of Ecology **17**:305-314.

Hook, D. D., M. A. Buford, and T. M. Williams. 1991. Impact of hurricane Hugo on the South Carolina coastal plain forest. Journal of Coastal Research **8**:291-300.

Hopkinson, C. S., A. E. Lugo, M. Alber, A. P. Covich, and S. J. Van Bloem. 2008. Forecasting effects of sea-level rise and windstorms on coastal and inland ecosystems. Frontiers in Ecology and the Environment **6**:255-263.

Hoppes, W. G. 1988. Seedfall pattern of several species of bird-dispersed plants in an Illinois woodland. Ecology **69**:320-329.

Horvitz, C. C., J. B. Pascarella, S. McMann, A. Freedman, and R. H. Hofstetter. 1998. Functional roles of invasive non-indigenous plants in hurricane-affected subtropical hardwood forests. Ecological Applications **8**:947-974.

Howard, J. J. 2012. Hurricane Katrina impact on a leveed bottomland hardwood forest in Louisiana. American Midland Naturalist **168**:56-69.

Huston, M. 1979. A general hypothesis of species diversity. The American Naturalist **113**:81-101.

IPCC, editor. 2007a. Climate Change 2007: Impacts, adaptation, and vulnerability. Contribution of working group II to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom.

IPCC. 2007b. Summary for Policymakers. In: Climate Change 2007: The physical science basis. Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Jasper, D. A., L. K. Abbott, and A. D. Robson. 1989. Soil disturbance reduces the infectivity of external hyphae of vesicular arbuscular mycorrhizal fungi. New Phytologist **112**:93-99.

Jones, M. D., D. M. Durall, and J. W. G. Cairney. 2003. Ectomycorrhizal fungal communities in young forest stands regenerating after clearcut logging. New Phytologist **157**:399-422.

Jones, R. H. and K. W. McLeod. 1989. Shade tolerance in seedlings of Chinese tallow tree, American sycamore, and cherrybark oak. Bulletin of the Torrey Botanical Club **116**:371-377.

Jubinsky, G. and L. C. Anderson. 1996. The invasive potential of Chinese tallow-tree (*Sapium sebiferum* Roxb.) in the southeast. Castanea **61**:226-231.

Juniper, S. and L. Abbott. 1993. Vesicular-arbuscular mycorrhizas and soil-salinity. Mycorrhiza **4**:45-57.

Kandalepas, D., K. J. Stevens, G. P. Shaffer, and W. J. Platt. 2010. How abundant are root-colonizing fungi in southeastern Louisiana's degraded marshes? Wetlands **30**:189-199.

Keane, R. M. and M. J. Crawley. 2002. Exotic plant invasions and the enemy release hypothesis. Trends in Ecology & Evolution **17**:164-170.

Keay, J., W. E. Rogers, R. Lankau, and E. Siemann. 2000. The role of allelopathy in the invasion of the Chinese tallow tree (*Sapium sebiferum*). Texas Journal of Science **52**:57-64.

Keeley, J. E., D. Lubin, and C. J. Fotheringham. 2003. Fire and grazing impacts on plant diversity and alien plant invasions in the southern Sierra Nevada. Ecological Applications **13**:1355-1374.

Kelly, A. E. and M. L. Goulden. 2008. Rapid shifts in plant distribution with recent climate change. Proceedings of the National Academy of Sciences of the United States of America **105**:11823-11826.

Khan, M. A. and I. A. Ungar. 1984. The effect of salinity and temperature on the germination of polymorphic seed and growth of *Atriplex triangularis* willd. American Journal of Botany **71**:481-489.

Klanderud, K. 2008. Species-specific responses of an alpine plant community under simulated environmental change. Journal of Vegetation Science **19**:363-372.

Klironomos, J., M. Zobel, M. Tibbett, W. D. Stock, M. C. Rillig, J. L. Parrent, M. Moora, A. M. Koch, J. M. Facelli, E. Facelli, I. A. Dickie, and J. D. Bever. 2011. Forces that structure plant communities: quantifying the importance of the mycorrhizal symbiosis. New Phytologist **189**:366-370.

Klironomos, J. N. 2002. Feedback with soil biota contributes to plant rarity and invasiveness in communities. Nature **417**:67-70.

Koca, D., B. Smith, and M. T. Sykes. 2006. Modelling regional climate change effects on potential natural ecosystems in Sweden. Climatic Change **78**:381-406.

Kozlowski, T. T. 1997. Response of woody plants to flooding and salinity. Tree Physiology Monograph No.1, Heron publishing, Victoria, Canada.

Krauss, K. W., J. L. Chambers, and J. A. Allen. 1998. Salinity effects and differential germination of several half-sib families of baldcypress from different seed sources. New Forests **15**:53-68.

Krischik, V. A. and R. F. Denno. 1990. Differences in environmental response between the sexes of the dioecious shrub, *Baccharis halimifolia* (Compositae). Oecologia **83**:176-181.

Kuhn, N. L. and J. B. Zedler. 1997. Differential effects of salinity and soil saturation on native and exotic plants of a coastal salt marsh. Estuaries **20**:391-403.

Kupfer, J. A., A. T. Myers, S. E. McLane, and G. N. Melton. 2008. Patterns of forest damage in a southern Mississippi landscape caused by hurricane Katrina. Ecosystems **11**:45-60.

Lapointe, M. and J. Brisson. 2012. A comparison of invasive *Acer platanoides* and native *A. saccharum* first-year seedlings: growth, biomass distribution and the influence of ecological factors in a forest understory. Forests **3**:190-206.

Lau, J. A., K. P. Puliafico, J. A. Kopshever, H. Steltzer, E. P. Jarvis, M. Schwarzlander, S. Y. Strauss, and R. A. Hufbauer. 2008. Inference of allelopathy is complicated by effects of activated carbon on plant growth. New Phytologist **178**:412-423.

Laurance, W. F. and T. J. Curran. 2008. Impacts of wind disturbance on fragmented tropical forests: A review and synthesis. Austral Ecology **33**:399-408.

Leigh, J., A. Hodge, and A. H. Fitter. 2009. Arbuscular mycorrhizal fungi can transfer substantial amounts of nitrogen to their host plant from organic material. New Phytologist **181**:199-207.

Lloret, F., A. Escudero, J. Maria Iriondo, J. Martinez-Vilalta, and F. Valladares. 2012. Extreme climatic events and vegetation: the role of stabilizing processes. Global Change Biology **18**:797-805.

Lodge, D. J. and W. H. McDowell. 1991. Summary of ecosystem-level effects of Caribbean hurricanes. Biotropica **23**:373-378.

Lugo, A. E. 2000. Effects and outcomes of Caribbean hurricanes in a climate change scenario. Science of the Total Environment **262**:243-251.

Lugo, A. E. 2008. Visible and invisible effects of hurricanes on forest ecosystems: an international review. Austral Ecology **33**:368-398.

Lynch, R. L., H. J. Chen, L. A. Brandt, and F. J. Mazzotti. 2009. Old world climbing fern (*Lygodium microphyllum*) invasion in hurricane caused treefalls. Natural Areas Journal **29**:210-215.

Macauley, J., L. M. Smith, V. Engle, L. Harwell, W. Berry, J. Hyland, M. Fulton, G. Lauenstein, P. Bourgeois, and T. Heitmuller. 2007. Environmental conditions in northern Gulf of Mexico coastal waters following hurricane Katrina. EPA/600/R-07/063. U.S. Environmental Protection Agency, office of research and development, national health and environmental effects research laboratory, Gulf ecology division, Gulf Breeze, Florida.

Mack, R. N., D. Simberloff, W. M. Lonsdale, H. Evans, M. Clout, and F. A. Bazzaz. 2000. Biotic invasions: Causes, epidemiology, global consequences, and control. Ecological Applications **10**:689-710.

Mangla, S., Inderjit, and R. M. Callaway. 2008. Exotic invasive plant accumulates native soil pathogens which inhibit native plants. Journal of Ecology **96**:58-67.

Marler, M. J., C. A. Zabinski, and R. M. Callaway. 1999a. Mycorrhizae indirectly enhance competitive effects of an invasive forb on a native bunchgrass. Ecology **80**:1180-1186.

Marler, M. J., C. A. Zabinski, T. Wojtowicz, and R. M. Callaway. 1999b. Mycorrhizae and fine root dynamics of *Centaurea maculosa* and native bunchgrasses in western Montana. Northwest Science **73**:217-224.

Martin, C. O. and S. P. Mott. 1997. "Youpon (*Ilex vomitoria*): Section 7.5.10, U.S. army corps of engineers wildlife resources management manual". Technical report EL-97-16, U.S. Army engineer waterways experiment station, Vicksburg, MS.

Martinez, M. L., A. Intralawan, G. Vazquez, O. Perez-Maqueo, P. Sutton, and R. Landgrave. 2007. The coasts of our world: Ecological, economic and social importance. Ecological Economics **63**:254-272.

Matlack, G. R. 2002. Exotic plant species in Mississippi, USA: Critical issues in management and research. Natural Areas Journal **22**:241-247.

Meekins, J. F. and B. C. McCarthy. 2001. Effect of environmental variation on the invasive success of a nonindigenous forest herb. Ecological Applications **11**:1336-1348.

McGonigle, T. P., M. H. Miller, D. G. Evans, G. L. Fairchild, and J. A. Swan. 1990. A new method which gives an objective measure of colonization of roots by vesicular-arbuscular mycorrhizal fungi. New Phytologist **115**:495-501.

Meehl, G. A., T. F. Stocker, W. D. Collins, P. Friedlingstein, A. T. Gaye, J. M. Gregory, A. Kitoh, R. Knutti, J. M. Murphy, A. Noda, S. C. B. Raper, I. G. Watterson, A. J. Weaver, and Z.-C. Zhao. 2007. Global Climate Projections.*in* S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller, editor. Climate change 2007: The physical science basis. Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Meier, M. F., M. B. Dyurgerov, U. K. Rick, S. O'Neel, W. T. Pfeffer, R. S. Anderson, S. P. Anderson, and A. F. Glazovsky. 2007. Glaciers dominate Eustatic sea-level rise in the 21st century. Science **317**:1064-1067.

Merry, K., P. Bettinger, and J. Hepinstall. 2009. Physical and biological responses of forests to tropical cyclones affecting the United States Atlantic Ocean and Gulf of Mexico coasts. American Journal of Environmental Sciences **5**:784-800.

Michener, W. K., E. R. Blood, K. L. Bildstein, M. M. Brinson, and L. R. Gardner. 1997. Climate change, hurricanes and tropical storms, and rising sea level in coastal wetlands. Ecological Applications **7**:770-801.

Middleton, B. A. 2009. Effects of hurricane Katrina on the forest structure of *Taxodium distichum* swamps of the Gulf coast USA. Wetlands **29**:80-87.

Mihulka, S., P. Pyšek, and J. Martínková. 2003. Invasiveness of Oenothera congeners in Europe related to their seedcharacteristics. – In: Child L. E., Brock J. H., Brundu G., Prach K., Pyšek P., Wade P. M. & Williamson M.(eds), Plant invasions: ecological threats and management solutions, p. 213–225, Backhuys Publishers,Leiden.

Miller, A. D., S. H. Roxburgh, and K. Shea. 2011. How frequency and intensity shape diversity-disturbance relationships. Proceedings of the National Academy of Sciences of the United States of America **108**:5643-5648

Miller, D. L., M. Thetford, and M. Schneider. 2008. Distance from the Gulf influences survival and growth of three barrier island dune plants. Journal of Coastal Research **24**:261-266.

Mitchell, C. E. and A. G. Power. 2003. Release of invasive plants from fungal and viral pathogens. Nature **421**:625-627.

Mitsch, W. J. A. and J. G. Gosselink. 2007. Wetlands. 4th edition edition. John Wiley and Sons, New York.

Mok, H.-F., S. K. Arndt, and C. R. Nitschke. 2012. Modelling the potential impact of climate variability and change on species regeneration potential in the temperate forests of South-Eastern Australia. Global Change Biology **18**:1053-1072.

Moon, D. C. and P. Stiling. 2004. The influence of a salinity and nutrient gradient on coastal vs. upland tritrophic complexes. Ecology **85**:2709-2716.

Moorhead, K. K. and M. M. Brinson. 1995. Response of wetlands to rising sea-level in the lower coastal-plain of North Carolina. Ecological Applications **5**:261-271.

Mousavi, M. E., J. L. Irish, A. E. Frey, F. Olivera, and B. L. Edge. 2011. Global warming and hurricanes: the potential impact of hurricane intensification and sea level rise on coastal flooding. Climatic Change **104**:575-597.

Mukerji, K. G. and A. Kapoor. 1986. Occurrence and importance of vesicular arbuscular mycorrhizal fungi in semiarid regions of India. Forest Ecology and Management **16**:117-126.

Murphy, H. T., B. D. Hardesty, C. S. Fletcher, D. J. Metcalfe, D. A. Westcott, and S. J. Brooks. 2008a. Predicting dispersal and recruitment of *Miconia calvescens* (Melastomataceae) in Australian tropical rainforests. Biological Invasions **10**:925-936.

Murphy, H. T., D. J. Metcalfe, M. G. Bradford, A. F. Ford, K. E. Galway, T. A. Sydes, and D. J. Westcott. 2008b. Recruitment dynamics of invasive species in rainforest habitats following Cyclone Larry. Austral Ecology **33**:495-502.

Myers, R. K. and D. H. van Lear. 1998. Hurricane-fire interactions in coastal forests of the south: a review and hypothesis. Forest Ecology and Management **103**:265-276.

Newsham, K. K., A. H. Fitter, and A. R. Watkinson. 1995. Arbuscular mycorrhiza protect an annual grass from root pathogenic fungi in the field. Journal of Ecology **83**:991-1000.

Neyland, R. and H. A. Meyer. 1997. Species diversity of Louisiana chenier woody vegetation remnants. Journal of the Torrey Botanical Society **124**:254-261.

Nicholls, R. J. and A. Cazenave. 2010. Sea-level rise and its impact on coastal zones. Science **328**:1517-1520.

Nijjer, S., R. A. Lankau, W. E. Rogers, and E. Siemann. 2002. Effects of temperature and light on Chinese tallow (*Sapium sebiferum*) and Texas sugarberry (*Celtis laevigata*) seed germination. Texas Journal of Science **54**:63-68.

Nijjer, S., W. E. Rogers, C. T. A. Lee, and E. Siemann. 2008. The effects of soil biota and fertilization on the success of *Sapium sebiferum*. Applied Soil Ecology **38**:1-11.

Ning, Z. H., R. E. Turner, T. Doyle, and K. K. Abdollahi. 2003. Preparing for a changing climate: potential consequences of climate variability and change - Gulf coast region. GCRCC and LSU Graphic Services, Baton Rouge, LA.

NOAA. 2007. Grand Bay National Estuarine Research Reserve, Mississippi.*in* C. J. Cleveland, editor. Encyclopedia of earth. Environmental information coalition, national council for science and the environment, Washington, D.C.

Otvos, E. G. 2007. Climate and weather of coastal Mississippi. Pages 22-46 *in* M. S. Peterson, G. L. Waggy, and M. S. Woodrey, editors. Grand Bay National Estuarine Research Reserve: An ecological characterization. Grand Bay National Estuarine Research Reserve, Moss Point, Mississippi.

Overpeck, J. T., L. C. Peterson, N. Kipp, J. Imbrie, and D. Rind. 1989. Climate change in the circum-north Atlantic region during the last deglaciation. Nature **338**:553-557.

Overpeck, J. T., D. Rind, and R. Goldberg. 1990. Climate-induced changes in forest disturbance and vegetation. Nature **343**:51-53.

Packer, A. and K. Clay. 2000. Soil pathogens and spatial patterns of seedling mortality in a temperate tree. Nature **404**:278-281.

Parendes, L. A. and J. A. Jones. 2000. Role of light availability and dispersal in exotic plant invasion along roads and streams in the H. J. Andrews Experimental Forest, Oregon. Conservation Biology **14**:64-75.

Parmesan, C. 1996. Climate and species' range. Nature **382**:765-766.

Parmesan, C. 2006. Ecological and evolutionary responses to recent climate change. Annual Review of Ecology Evolution and Systematics **37**:637-669.

Parmesan, C., N. Ryrholm, C. Stefanescu, J. K. Hill, C. D. Thomas, H. Descimon, B. Huntley, L. Kaila, J. Kullberg, T. Tammaru, W. J. Tennent, J. A. Thomas, M. Warren, and Rr. 1999. Poleward shifts in geographical ranges of butterfly species associated with regional warming. Nature **399**:579-583.

Parmesan, C. and G. Yohe. 2003. A globally coherent fingerprint of climate change impacts across natural systems. Nature **421**:37-42.

Pattison, R. R. and R. N. Mack. 2008. Potential distribution of the invasive tree *Triadica sebifera* (Euphorbiaceae) in the United States: evaluating CLIMEX predictions with field trials. Global Change Biology **14**:813-826.

Penfound, W. T. and E. S. Hathaway. 1938. Plant communities in the marshlands of southeastern Louisiana. Ecological Monographs **8**:3-56.

Perry, L. G., D. C. Andersen, L. V. Reynolds, S. M. Nelson, and P. B. Shafroth. 2012. Vulnerability of riparian ecosystems to elevated CO_2 and climate change in arid and semiarid western North America. Global Change Biology **18**:821-842.

Peterson, C. J. and S. T. A. Pickett. 1995. Forest reorganization- a case-study in an old-growth forest catastrophic blowdown. Ecology **76**:763-774.

Peterson, M. S., G. L. Waggy, and M. S. Woodrey. 2007. Grand Bay National Estuarine Research Reserve: An ecological characterization. Grand Bay National Estuarine Research Reserve, Moss Point, Mississippi. 268 pp.

Pickett, S. T. A. and P. S. White, editors. 1985. The ecology of natural disturbance and patch dynamics. Academic Press, Orlando, Florida.

Pimentel, D., L. Lach, R. Zuniga, and D. Morrison. 2000. Environmental and economic costs of nonindigenous species in the United States. Bioscience **50**:53-65.

Pimentel, D., R. Zuniga, and D. Morrison. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics **52**:273-288.

Porter, R. H., M. Durrell, and H. J. Romm. 1947. The use of 2, 3, 5-triphenyl-tetrazoliumchloride as a measure of seed germinability. Plant Physiology **22**:149–159.

Pringle, A., J. D. Bever, M. Gardes, J. L. Parrent, M. C. Rillig, and J. N. Klironomos. 2009. Mycorrhizal symbioses and plant invasions. Annual Review of Ecology Evolution and Systematics **40**:699-715.

Rahmstorf, S. 2007. A semi-empirical approach to projecting future sea-level rise. Science **315**:368-370.

Randall, J. M. and J. Marinelli. 1996. Invasive plants, weeds of the global garden. Brooklyn Botanic Garden, Brooklyn, New York.

Reinhardt, D. R. and R. M. Miller. 1990. Size classes of root diameter and mycorrhizal fungal colonization in 2 temperate grassland communities. New Phytologist **116**:129-136.

Reinhart, K. O. and R. M. Callaway. 2004. Soil biota facilitate exotic Acer invasions in Europe and North America. Ecological Applications **14**:1737-1745.

Reinhart, K. O. and R. M. Callaway. 2006. Soil biota and invasive plants. New Phytologist **170**:445-457.

Reinhart, K. O., A. Packer, W. H. Van der Putten, and K. Clay. 2003. Plant-soil biota interactions and spatial distribution of black cherry in its native and invasive ranges. Ecology Letters **6**:1046-1050.

Reinhart, K. O., T. Tytgat, W. H. Van der Putten, and K. Clay. 2010. Virulence of soil-borne pathogens and invasion by *Prunus serotina*. New Phytologist **186**:484-495.

Rejmanek, M. 1989. Invasibility of plant communities. Pages 369-388 *in* J. A. Drake, H. A. Mooney, F. di Castri, I. H. Groves, F. J. Kruger, M. Rejmanek, and M. Williamson, editors. Biological invasions: a global perspective. Wiley, Chichester, England.

Rejmanek, M. 2000. Invasive plants: approaches and predictions. Austral Ecology **25**:497-506.

Rejmanek, M. and D. M. Richardson. 1996. What attributes make some plant species more invasive? Ecology **77**:1655-1661.

Renne, I. J., W. C. Barrow, L. A. J. Randall, and W. C. Bridges. 2002. Generalized avian dispersal syndrome contributes to Chinese tallow tree (*Sapium sebiferum*, Euphorbiaceae) invasiveness. Diversity and Distributions **8**:285-295.

Renne, I. J., S. A. Gauthreaux, and C. A. Gresham. 2000. Seed dispersal of the Chinese tallow tree (*Sapium sebiferum* (L.) Roxb.) by birds in coastal South Carolina. American Midland Naturalist **144**:202-215.

Renne, I. J., T. P. Spira, and W. C. Bridges. 2001. Effects of habitat, burial, age and passage through birds on germination and establishment of Chinese tallow tree in coastal South Carolina. Journal of the Torrey Botanical Society **128**:109-119.

Rey, P. J. and J. M. Alcantara. 2000. Recruitment dynamics of a fleshy-fruited plant (*Olea europaea*): connecting patterns of seed dispersal to seedling establishment. Journal of Ecology **88**:622-633.

Reynolds, H. L., A. Packer, J. D. Bever, and K. Clay. 2003. Grassroots ecology: Plant-microbe-soil interactions as drivers of plant community structure and dynamics. Ecology **84**:2281-2291.

Rhoades, J. D. 1996. Salinity: Electrical conductivity and total dissolved solids. Pages 417–435 *in* D. L. Sparks, A. L. Page, P. A. Helmuke, R. H. Loeppert, P. N. Soltanpour, M. A. Tabatabai, C. T. Johnston, and M. E. Sumner, editors. Methods of soil analysis. Part 3. Chemical methods. SSSA Book Ser. 5. SSSA, Madison, WI.

Richards, C. L., O. Bossdorf, N. Z. Muth, J. Gurevitch, and M. Pigliucci. 2006. Jack of all trades, master of some? On the role of phenotypic plasticity in plant invasions. Ecology Letters **9**:981-993.

Richards, C. L., R. L. Walls, J. P. Bailey, R. Parameswaran, T. George, and M. Pigliucci. 2008. Plasticity in salt tolerance traits allows for invasion of novel habitat by Japanese knotweed s. l. (*Fallopia japonica* and *F. bohemica*, Polygonaceae). American Journal of Botany **95**:931-942.

Richardson, D. M., N. Allsopp, C. M. D'Antonio, S. J. Milton, and M. Rejmanek. 2000a. Plant invasions - the role of mutualisms. Biological Reviews **75**:65-93.

Richardson, D. M., P. Pyšek, M. Rejmánek, M. G. Barbour, F. D. Panetta, and C. J. West. 2000b. Naturalization and invasions of alien plants: concepts and definitions. Diversity and Distribution **6**:93–107.

Rickerl, D. H., F. O. Sancho, and S. Ananth. 1994. Vesicular-arbuscular endomycorrhizal colonization of wetland plants. Journal of Environmental Quality **23**:913-916.

Roberts, K. J. and R. C. Anderson. 2001. Effect of garlic mustard [*Alliaria petiolata* (Beib. Cavara & Grande)] extracts on plants and arbuscular mycorrhizal (AM) fungi. American Midland Naturalist **146**:146-152.

Rogers, W. E. and E. Siemann. 2003. Effects of simulated herbivory and resources on Chinese tallow tree (*Sapium sebiferum*, Euphorbiaceae) invasion of native coastal prairie. American Journal of Botany **90**:243-249.

Root, T. L., J. T. Price, K. R. Hall, S. H. Schneider, C. Rosenzweig, and J. A. Pounds. 2003. Fingerprints of global warming on wild animals and plants. Nature **421**:57-60.

Saha, A. K., S. Saha, J. Sadle, J. Jiang, M. S. Ross, R. M. Price, L. S. L. O. Sternberg, and K. S. Wendelberger. 2011. Sea level rise and South Florida coastal forests. Climatic Change **107**:81-108.

SAS. 2008. SAS/STAT user's guide, version 9.2. Cary, NC. SAS Institute.

Scavia, D., J. C. Field, D. F. Boesch, R. W. Buddemeier, V. Burkett, D. R. Cayan, M. Fogarty, M. A. Harwell, R. W. Howarth, C. Mason, D. J. Reed, T. C. Royer, A. H. Sallenger, and J. G. Titus. 2002. Climate change impacts on US coastal and marine ecosystems. Estuaries **25**:149-164.

Scheffer, M., S. Carpenter, J. A. Foley, C. Folke, and B. Walker. 2001. Catastrophic shifts in ecosystems. Nature **413**:591-596.

Schmidt, S. K. and K. M. Scow. 1986. Mycorrhizal fungi on the Galaoagos Islands. Biotropica **18**:236-240.

Semones, S. W. and D. R. Young. 1995. Vam association in the shrub *Myrica cerifera* on a Virginia, USA barrier island. Mycorrhiza **5**:423-429.

Shaffer, G. P., W. B. Wood, S. S. Hoeppner, T. E. Perkins, J. Zoller, and D. Kandalepas. 2009. Degradation of Baldcypress–Water Tupelo swamp to marsh and open water in southeastern Louisiana, U.S.A.: An irreversible trajectory? Journal of Coastal Research **54**:152-165.

Shea, K. and P. Chesson. 2002. Community ecology theory as a framework for biological invasions. Trends in Ecology & Evolution **17**:170-176.

Sheldrick, B. H. and C. Wang. 1993. Particle size distribution. In Soil sampling and methods of soil analysis. Edited by M.R. Carter. Lewis Publishers, Inc., Boca Raton, Fla.

Sheppard, C. S., J. M. Alexander, and R. Billeter. 2012. The invasion of plant communities following extreme weather events under ambient and elevated temperature. Plant Ecology **213**:1289-1301.

Shiels, A. B., J. K. Zimmerman, D. C. Garcia-Montiel, I. Jonckheere, J. Holm, D. Horton, and N. Brokaw. 2010. Plant responses to simulated hurricane impacts in a subtropical wet forest, Puerto Rico. Journal of Ecology **98**:659-673.

Siemann, E., W. E. Rogers, and S. J. Dewalt. 2006. Rapid adaptation of insect herbivores to an invasive plant. Proceedings of the Royal Society B-Biological Sciences **273**:2763-2769.

Silliman, B. R. and M. D. Bertness. 2004. Shoreline development drives invasion of *Phragmites australis* and the loss of plant diversity on New England salt marshes. Conservation Biology **18**:1424-1434.

Sims-Chilton, N. M., M. P. Zalucki, and Y. M. Buckley. 2010. Long term climate effects are confounded with the biological control programme against the invasive weed *Baccharis halimifolia* in Australia. Biological Invasions **12**:3145-3155.

Smith, L. L., A. DiTommaso, J. Lehmann, and S. Greipsson. 2008. Effects of arbuscular mycorrhizal fungi on the exotic invasive vine Pale Swallow-Wort (*Vincetoxicum rossicum*). Invasive Plant Science and Management **1**:142-152.

Smith, S. D., T. E. Huxman, S. F. Zitzer, T. N. Charlet, D. C. Housman, J. S. Coleman, L. K. Fenstermaker, J. R. Seemann, and R. S. Nowak. 2000. Elevated CO_2 increases productivity and invasive species success in an arid ecosystem. Nature **408**:79-82.

Smith, S. E. and D. J. Read. 1997. Mycorrhizal Symbiosis. Academic Press, San Diego, CA.

Snitzer, J. L., D. H. Boucher, and K. L. Kyde. 2005. Response of exotic invasive plant species to forest damage caused by Hurricane Isabel. Pp 209–214. *In* Sellner, K. G., editor. (Ed.).

Hurricane Isabel in Perspective. Chesapeake Research Consortium, CRC Publication 05-160. Edgewater, MD.

Solomon, S., G.-K. Plattner, R. Knutti, and P. Friedlingstein. 2009. Irreversible climate change due to carbon dioxide emissions. Proceedings of the National Academy of Sciences of the United States of America **106**:1704-1709.

Stanturf, J. A., S. L. Goodrick, and K. W. Outcalt. 2007. Disturbance and coastal forests: A strategic approach to forest management in hurricane impact zones. Forest Ecology and Management **250**:119-135.

Stinson, K. A., S. A. Campbell, J. R. Powell, B. E. Wolfe, R. M. Callaway, G. C. Thelen, S. G. Hallett, D. Prati, and J. N. Klironomos. 2006. Invasive plant suppresses the growth of native tree seedlings by disrupting belowground mutualisms. Plos Biology **4**:727-731.

Stokstad, E. 2005. After Katrina - Louisiana's wetlands struggle for survival. Science **310**:1264-1266.

Sturm, M., C. Racine, and K. Tape. 2001. Climate change - Increasing shrub abundance in the Arctic. Nature **411**:546-547.

Sun, Z. K. and W. M. He. 2010. Evidence for enhanced mutualism hypothesis: *Solidago canadensis* plants from regular soils perform better. Plos One **5**.

Sutton, J. R., T. J. Stohlgren, and K. G. Beck. 2007. Predicting yellow toadflax infestations in the flat tops wilderness of colorado. Biological Invasions **9**:783-793.

Tate, A. S. and L. L. Battaglia. 2013. Community disassembly and reassembly following experimental storm surge and wrack application. Journal of Vegetation Science **24**:46-57.

Tavares, R. C., P. F. M. Filho, C. F. de Lacerda, and J. Silva. 2012. Mycorrhizal colonization and root nodulation in sabia seedlings (*Mimosa caesalpiniaefolia* Benth.) at different salinity levels. Revista Ciencia Agronomica **43**:409-416.

Theoharides, K. A. and J. S. Dukes. 2007. Plant invasion across space and time: factors affecting nonindigenous species success during four stages of invasion. New Phytologist **176**:256-273.

Thuiller, W. 2007. Biodiversity - Climate change and the ecologist. Nature **448**:550-552.

Thuiller, W., D. M. Richardson, and G. F. Midgley. 2007. Will climate change promote alien plant invasions? Pages 197-211 *in* W. Nentwig, editor. Biological Invasions. Springer-Verlag, Berlin Heidelberg.

Thuiller, W., D. M. Richardson, P. Pysek, G. F. Midgley, G. O. Hughes, and M. Rouget. 2005. Niche-based modelling as a tool for predicting the risk of alien plant invasions at a global scale. Global Change Biology **11**:2234-2250.

Titus, J. G., K. E. Anderson, D. R. Cahoon, D. B. Gesch, S. K. Gill, B. T. Gutierrez, E. R. E.R.Thieler, and S. J. Williams. 2009. Coastal sensitivity to sea-level rise: A focus on the Mid-Atlantic region. Synthesis and Assessment Product 4.1 Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research.

Tolliver, K. S., D. W. Martin, and D. R. Young. 1997. Freshwater and saltwater flooding response for woody species common to barrier island swales. Wetlands **17**:10-18.

Turner, M. G., W. L. Baker, C. J. Peterson, and R. K. Peet. 1998a. Factors influencing succession: Lessons from large, infrequent natural disturbances. Ecosystems **1**:511-523.

Turner, M. G. and V. H. Dale. 1998b. Comparing large, infrequent disturbances: What have we learned? Ecosystems **1**:493-496.

Turner, M. G., S. L. Collins, A. E. Lugo, J. J. Magnuson, T. S. Rupp, and F. J. Swanson. 2003. Disturbance dynamics and ecological response: The contribution of long-term ecological research. Bioscience **53**:46-56.

Ungar, I. A. 1978. Halophyte seed germination. Botanical Reviews **44**: 233-245.

Upton, G. J. G. 1992. Fisher exact test. Journal of the Royal Statistical Society Series a-Statistics in Society **155**:395-402.

van der Heijden, M. G. A., J. N. Klironomos, M. Ursic, P. Moutoglis, R. Streitwolf-Engel, T. Boller, A. Wiemken, and I. R. Sanders. 1998. Mycorrhizal fungal diversity determines plant biodiversity, ecosystem variability and productivity. Nature **396**:69-72.

Van der Putten, W. H., L. E. M. Vet, J. A. Harvey, and F. L. Wackers. 2001. Linking above- and belowground multitrophic interactions of plants, herbivores, pathogens, and their antagonists. Trends in Ecology & Evolution **16**:547-554.

Vargas, R., N. Hasselquist, E. B. Allen, and M. F. Allen. 2010. Effects of a hurricane disturbance on aboveground forest structure, arbuscular mycorrhizae and belowground carbon in a restored tropical forest. Ecosystems **13**:118-128.

Vasquez, E. A., E. P. Glenn, J. J. Brown, G. R. Guntenspergen, and S. G. Nelson. 2005. Salt tolerance underlies the cryptic invasion of North American salt marshes by an introduced haplotype of the common reed *Phragmites australis* (Poaceae). Marine Ecology-Progress Series **298**:1-8.

Vazquez, D. 2006. Exploring the relationship between niche breadth and invasion success. Pages 307–322 *in* M. W. Cadotte, S. M. S. M. Mc Mahon, and T. Fukami, editors. Conceptual ecology and invasions biology Springer, Dordrecht, The Netherlands.

Vermeer, M. and S. Rahmstorf. 2009. Global sea level linked to global temperature. Proceedings of the National Academy of Sciences of the United States of America **106**:21527-21532.

Vitousek, P. M., C. M. Dantonio, L. L. Loope, M. Rejmanek, and R. Westbrooks. 1997. Introduced species: A significant component of human-caused global change. New Zealand Journal of Ecology **21**:1-16.

Vogelsang, K. M. and J. D. Bever. 2009. Mycorrhizal densities decline in association with nonnative plants and contribute to plant invasion. Ecology **90**:399-407.

von der Lippe, M., J. M. Bullock, I. Kowarik, T. Knopp, and M. Wichmann. 2013. Human-mediated dispersal of seeds by the airflow of vehicles. PloS one **8**:e52733.

Von der Lippe, M. and I. Kowarik. 2007. Long-distance dispersal of plants by vehicles as a driver of plant invasions. Conservation Biology **21**:986-996.

Wall, D. H. and J. C. Moore. 1999. Interactions underground - Soil biodiversity, mutualism, and ecosystem processes. Bioscience **49**:109-117.

Wall, D. P. and S. P. Darwin. 1999. Vegetation and elevational gradients within a bottomland hardwood forest of southeastern Louisiana. American Midland Naturalist **142**:17-30.

Walls, R. L. 2010. Hybridization and plasticity contribute to divergence among coastal and wetland populations of invasive hybrid japanese knotweed s.l. (*Fallopia* spp.). Estuaries and Coasts **33**:902-918.

Walther, G. R. 2010. Community and ecosystem responses to recent climate change. Philosophical Transactions of the Royal Society B-Biological Sciences **365**:2019-2024.

Walther, G. R., E. Post, P. Convey, A. Menzel, C. Parmesan, T. J. C. Beebee, J. M. Fromentin, O. Hoegh-Guldberg, and F. Bairlein. 2002. Ecological responses to recent climate change. Nature **416**:389-395.

Wang, F. and Y. J. Xu. 2009. Hurricane Katrina-induced forest damage in relation to ecological factors at landscape scale. Environmental Monitoring and Assessment **156**:491-507.

Wang, H. H., W. E. Grant, T. M. Swannack, J. Gan, W. E. Rogers, T. E. Koralewski, J. H. Miller, and J. W. Taylor Jr. 2011. Predicted range expansion of Chinese tallow tree (*Triadica sebifera*) in forestlands of the southern United States. Diversity and Distribution **17**:552–565.

Wang, Q., C. H. Wang, B. Zhao, Z. J. Ma, Y. Q. Luo, J. K. Chen, and B. Li. 2006. Effects of growing conditions on the growth of and interactions between salt marsh plants: Implications for invasibility of habitats. Biological Invasions **8**:1547-1560.

Wang, W. X., B. Vinocur, and A. Altman. 2003. Plant responses to drought, salinity and extreme temperatures: towards genetic engineering for stress tolerance. Planta **218**:1-14.

Wardle, D. A. 2002. Communities and ecosystems: linking the aboveground and belowground components. Princeton University Press, Princeton, New Jersey, USA.

Wardle, D. A., R. D. Bardgett, J. N. Klironomos, H. Setala, W. H. van der Putten, and D. H. Wall. 2004. Ecological linkages between aboveground and belowground biota. Science **304**:1629-1633.

Weber, E. and C. M. D'Antonio. 1999. Germination and growth responses of hybridizing *Carpobrotus* species (Aizoaceae) from coastal California to soil salinity. American Journal of Botany **86**:1257-1263.

Webster, P. J., G. J. Holland, J. A. Curry, and H. R. Chang. 2005. Changes in tropical cyclone number, duration, and intensity in a warming environment. Science **309**:1844-1846.

Weisshuhn, K. and D. Prati. 2009. Activated carbon may have undesired side effects for testing allelopathy in invasive plants. Basic and Applied Ecology **10**:500-507.

Williams, K., K. C. Ewel, R. P. Stumpf, F. E. Putz, and T. W. Workman. 1999a. Sea-level rise and coastal forest retreat on the west coast of Florida, USA. Ecology **80**:2045-2063.

Williams, K., Z. S. Pinzon, R. P. Stumpf, and E. A. Raabe. 1999b. Sea-level rise and coastal forests on the Gulf of Mexico. Open-file report 99-441. U.S. Geological Survey, Center for Coastal Geology, St. Petersburg, Florida.

Williamson, M. 1996. Biological invasions. Chapman and Hall, London.

Wolfe, B. E. and J. N. Klironomos. 2005. Breaking new ground: Soil communities and exotic plant invasion. Bioscience **55**:477-487.

Wolfe, B. E., V. L. Rodgers, K. A. Stinson, and A. Pringle. 2008. The invasive plant *Alliaria petiolata* (garlic mustard) inhibits ectomycorrhizal fungi in its introduced range. Journal of Ecology **96**:777-783.

Woodell, S. R. J. 1985. Salinity and seed germination patterns in coastal plants. Vegetatio **61**: 223-229.

Young, D. R., D. L. Erickson, and S. W. Semones. 1994. Salinity and the small-scale distribution of 3 Barrier-Island shrubs. Canadian Journal of Botany-Revue Canadienne De Botanique **72**:1365-1372.

Young, D. R., G. F. Shao, and J. H. Porter. 1995. Spatial and temporal growth dynamics of barrier island shrub thickets. American Journal of Botany **82**:638-645.

Zabinski, C. A., L. Quinn, and R. M. Callaway. 2002. Phosphorus uptake, not carbon transfer, explains arbuscular mycorrhizal enhancement of *Centaurea maculosa* in the presence of native grassland species. Functional Ecology **16**:758-765.

Zar, J. H. 1999. Biostatistical analysis. 3rd edition. Prentice Hall.

Zavaleta, E. S. 2000. Valuing ecosystem services lost to *Tamarix* invasion in the United States. Pages 261–300 *in* H. A. a. H. Mooney, R.J., editor. Invasive species in a changing world. Island Press.

Zekri, M. 1993. Salinity and calcium effects on emergence, growth and mineral composition of seedlings of eight citrus rootstocks. Journal of Horticultural Science **68**: 53-62.

Zhang, Q., R. Yang, J. Tang, H. Yang, S. Hu, and X. Chen. 2010. Positive feedback between mycorrhizal fungi and plants influences plant invasion success and resistance to invasion. Plos One **5**:1-10.

Zimmerman, J. K., E. M. Everham, R. B. Waide, D. J. Lodge, C. M. Taylor, N. V. L. Brokaw, and Pv. 1994. Responses of tree species to hurricane winds in subtropical wet forest in Puerto-Rico - implications for tropical tree life-histories. Journal of Ecology **82**:911-922.

Ziska, L. H. 2003. Evaluation of the growth response of six invasive species to past, present and future atmospheric carbon dioxide. Journal of Experimental Botany **54**:395-404.

Zou, J. W., W. E. Rogers, and E. Siemann. 2009. Plasticity of *Sapium sebiferum* seedling growth to light and water resources: Inter- and intraspecific comparisons. Basic and Applied Ecology **10**:79-88.

APPENDICES

Appendix A. Raw data from observational study that assessed the probability of occurrence of invasive species *Triadica sebifera* and native species; *Baccharis halimifolia*, *Ilex vomitoria*, and *Morella cerifera* in Grand Bay National Estuarine Research Reserve (GBNERR). R = random, T = juvenile

Distance to coast (m)	Sampling point	Species	Pres/ abs	Soil water conductivity (mS/cm)	% canopy openness	% Sand	% Silt	% Clay	% Nitrogen	% Carbon	Carbon/Nitrogen
0	R	*B. halimifolia*	0	6.66	98.15	73.6	10.6	15.8	0.201	3.136	15.616
100	R	*B. halimifolia*	0	7.53	98.15	30.4	43	26.6	0.386	4.648	12.035
200	R	*B. halimifolia*	0	9.4	98.15	61.8	18.3	19.9	0.373	5.527	14.819
300	R	*B. halimifolia*	0	9.03	98.15	59.1	30.9	10	0.459	5.935	12.939
400	R	*B. halimifolia*	0	7.4	98.15	13.8	43.6	42.6	0.291	4.170	14.353
500	R	*B. halimifolia*	0	9.28	98.15	56.8	16.9	26.3	0.173	2.682	15.472
600	R	*B. halimifolia*	0	8.49	98.15	57.1	18.3	24.6	0.324	4.831	14.893
700	R	*B. halimifolia*	0	3.567	98.15	67	14	19	0.047	0.424	8.967
800	R	*B. halimifolia*	0	8.59	98.15	34.4	35	30.6	0.253	4.045	15.992
900	R	*B. halimifolia*	0	10.88	98.15	35	38.4	26.6	0.318	4.948	15.583
1000	R	*B. halimifolia*	0	7.38	98.15	69.5	18.6	11.9	0.280	4.108	14.654
1100	R	*B. halimifolia*	0	10.1	98.15	27	39	34	0.289	3.897	13.495
1200	R	*B. halimifolia*	0	11.53	98.15	.	.	.	0.451	6.415	14.211
1300	R	*B. halimifolia*	0	8.94	98.15	74.4	6.4	19.2	0.467	6.793	14.551
1400	R	*B. halimifolia*	0	6.43	98.15	59.5	24.3	16.2	0.215	2.749	12.772
1500	R	*B. halimifolia*	0	10	98.15	37	52	11	0.436	5.976	13.693
1600	R	*B. halimifolia*	0	8.34	98.15	73.2	12.3	14.5	0.411	6.091	14.814
1700	R	*B. halimifolia*	0	7.3	98.15	30.4	51	18.6	0.272	3.751	13.791
1800	R	*B. halimifolia*	0	9	98.15	29	49.7	21.3	0.317	4.648	14.678
1900	R	*B. halimifolia*	0	8	98.15	43	37	20	0.282	4.293	15.245
2000	R	*B. halimifolia*	0	11.7	98.15	32.4	27.2	40.4	0.381	5.262	13.821
2100	R	*B. halimifolia*	0	8.75	98.15	67.6	16.3	16.1	0.154	2.002	12.975
2200	R	*B. halimifolia*	0	11.63	98.15	13.8	26.4	59.8	0.284	3.950	13.903
2300	R	*B. halimifolia*	0	5.7	98.15	39.4	36.3	24.3	0.095	1.060	11.213
2400	R	*B. halimifolia*	0	8.725	98.15	67.1	10.3	22.6	0.375	6.057	16.170
2500	R	*B. halimifolia*	0	10.28	98.15	65.1	15.4	19.5	0.312	5.125	16.407
2600	R	*B. halimifolia*	0	9.76	98.15	59.5	21.2	19.3	0.324	4.831	14.893
2700	R	*B. halimifolia*	0	9.66	98.15	70.9	13.2	15.9	0.386	5.829	15.113
2800	R	*B. halimifolia*	0	10.5	98.15	67.2	11.5	21.3	0.345	5.384	15.610
2900	R	*B. halimifolia*	0	5.76	98.15	59.5	18.9	21.6	0.217	2.588	11.923
3000	R	*B. halimifolia*	0	9.19	98.15	39.6	39	21.4	0.389	5.392	13.861

Appendix A. Raw data from observational study, continued

3100	R	*B. halimifolia*	0	7.91	98.15	64.1	22	13.9	0.364	5.916	16.244
3200	R	*B. halimifolia*	0	8.5	98.15	57.5	20.6	21.9	0.263	3.985	15.147
3300	R	*B. halimifolia*	0	9.3	98.15	45.8	19.6	34.6	0.281	4.065	14.476
3400	R	*B. halimifolia*	0	7.11	98.15	60.5	31.2	8.3	0.433	6.931	15.991
3500	R	*B. halimifolia*	0	7.1	98.15	64.1	18.6	17.3	0.446	6.784	15.200
3600	R	*B. halimifolia*	0	9.668	98.15	49.1	40.6	10.3	0.472	6.799	14.396
3700	R	*B. halimifolia*	0	8.72	98.15	39	33.2	27.8	0.208	2.956	14.241
3800	R	*B. halimifolia*	0	13	98.15	67.5	11.2	21.3	0.418	7.770	18.566
3900	R	*B. halimifolia*	0	15.43	98.15	48.9	39.8	11.3	0.464	9.034	19.483
4000	R	*B. halimifolia*	0	14.6	98.15	59.5	25.5	15	1.079	11.161	10.342
4100	R	*B. halimifolia*	0	18.46	98.15	56.8	29.9	13.3	0.482	8.177	16.957
4200	R	*B. halimifolia*	0	16.25	97.87	68.8	19.3	11.9	0.420	9.002	21.415
4300	R	*B. halimifolia*	0	6.07	97.69	59.9	18.9	21.2	0.151	3.122	20.669
4400	R	*B. halimifolia*	0	3.93	90.64	51.6	28	20.4	0.079	1.297	16.380
4500	R	*B. halimifolia*	1	3.9	89.67	39.2	42.7	18.1	0.060	0.240	4.003
4600	R	*B. halimifolia*	0	4.72	88.53	37	35.2	27.8	0.148	1.914	12.927
4625	R	*B. halimifolia*	0	3.891	85.58	52.2	22.3	25.5	0.071	1.055	14.817
4650	R	*B. halimifolia*	0	0.892	73.73	57.3	26.3	16.4	0.068	0.899	13.266
4650	T	*B. halimifolia*	1	1.765	78.58	55.6	20.3	24.1	0.085	1.203	14.240
4675	R	*B. halimifolia*	0	0.4494	60.46	81.5	7.2	11.3	0.068	1.037	15.159
4675	T	*B. halimifolia*	1	0.7	64.12	77.3	12.6	10.1	0.054	0.955	17.589
4700	T	*B. halimifolia*	1	0.476	48.79	75.8	8.2	16	0.109	1.046	9.573
4700	R	*B. halimifolia*	0	0.463	59.45	63.3	14.3	22.4	0.091	1.229	13.512
4725	R	*B. halimifolia*	0	0.2	58.7	57.9	24.6	17.5	0.106	0.957	9.032
4725	T	*B. halimifolia*	1	0.3418	56.07	69.5	2.6	27.9	0.050	0.793	15.959
4750	R	*B. halimifolia*	0	0.3425	43.58	35.8	36	28.2	0.083	0.702	8.454
4750	T	*B. halimifolia*	1	0.3592	58	65.3	16	18.7	0.115	1.725	15.036
4775	T	*B. halimifolia*	1	0.456	47.58	75.9	12	12.1	0.118	1.716	14.504
4775	R	*B. halimifolia*	0	0.2601	40.31	59	20.3	20.7	0.149	2.775	18.569
4800	R	*B. halimifolia*	0	0.3266	40.16	76.4	11.8	11.8	0.053	0.694	13.180
4800	T	*B. halimifolia*	1	0.2882	42.4	74.1	0	25.9	0.099	1.614	16.312
4825	T	*B. halimifolia*	1	0.499	45.07	75.9	9.7	14.4	0.060	0.605	10.169
4825	R	*B. halimifolia*	0	2.652	48.08	49	22	29	0.060	0.967	16.123
4850	R	*B. halimifolia*	1	1.354	71.21	51	21.7	27.3	0.127	1.887	14.864
4850	T	*B. halimifolia*	1	0.4532	51.09	58.2	24.3	17.5	0.069	1.147	16.672
4875	R	*B. halimifolia*	0	1.5774	79.52	54.4	28.5	17.1	0.114	1.741	15.319

Appendix A. Raw data from observational study, continued

4875	T	*B. halimifolia*	1	1.828	72.62	57.5	26.4	16.1	0.163	3.193	19.550	
4900	T	*B. halimifolia*	1	2.48	81.39	52.5	27.5	20	0.096	1.727	18.037	
4900	R	*B. halimifolia*	0	3.735	85.12	10.2	68	21.8	0.181	3.492	19.271	
4925	R	*B. halimifolia*	0	4.35	84.05	64.7	17.8	17.5	0.379	6.877	18.142	
4950	R	*B. halimifolia*	0	4.75	81.5	77.3	13.5	9.2	0.556	8.788	15.808	
4975	R	*B. halimifolia*	0	2.28	83.92	52.7	19.7	27.6	0.159	3.044	19.125	
4975	T	*B. halimifolia*	1	3.52	86.26	71.3	17.2	11.5	0.650	12.449	19.164	
5000	R	*B. halimifolia*	0	1.012	84.23	79.5	3.2	17.3	0.153	2.412	15.736	
5000	T	*B. halimifolia*	1	2.621	87.21	57.3	20.3	22.4	0.184	3.038	16.501	
5025	R	*B. halimifolia*	0	2.025	85.44	59	20.9	20.1	0.103	1.951	18.928	
5025	T	*B. halimifolia*	1	1.752	84.93	70.9	14.4	14.7	0.129	3.010	23.368	
5050	R	*B. halimifolia*	0	1.4	79.42	56.1	20.6	23.3	0.174	2.207	12.677	
5050	T	*B. halimifolia*	1	1.84	81.43	44.7	30.6	24.7	0.093	1.811	19.412	
5075	R	*B. halimifolia*	0	1.444	71.6	53	20.9	26.1	0.104	1.477	14.220	
5075	T	*B. halimifolia*	1	1.851	75.33	57	20.9	22.1	0.189	2.831	14.956	
5100	T	*B. halimifolia*	1	0.691	53.44	69.6	8.3	22.1	0.087	1.050	12.110	
5100	R	*B. halimifolia*	0	0.4307	48.14	61.6	14.3	24.1	0.105	1.699	16.182	
5125	R	*B. halimifolia*	0	0.332	48.52	59	14	27	0.123	1.952	15.829	
5150	T	*B. halimifolia*	1	0.45	39.98	63.9	18	18.1	0.096	1.190	12.408	
5150	R	*B. halimifolia*	0	0.547	46.11	59.6	18.3	22.1	0.124	1.595	12.843	
5175	R	*B. halimifolia*	0	0.4485	42.44	68.4	16.6	15	0.076	1.050	13.791	
5175	T	*B. halimifolia*	1	0.58	42.39	65.9	8.3	25.8	0.054	0.770	14.309	
5200	T	*B. halimifolia*	1	0.325	38.49	76.1	8.6	15.3	0.062	0.921	14.928	
5200	R	*B. halimifolia*	0	0.426	40.58	65	16.3	18.7	0.062	1.059	17.190	
5225	R	*B. halimifolia*	0	1.601	70.58	45	18	37	0.244	3.834	15.703	
5225	T	*B. halimifolia*	1	0.4459	60.08	53.5	16.9	29.6	0.172	2.843	16.513	
5250	R	*B. halimifolia*	0	1.788	70.47	59.6	18.3	22.1	0.243	2.772	11.408	
5275	R	*B. halimifolia*	0	0.2601	39.15	65	12.3	22.7	0.130	2.107	16.204	
5300	T	*B. halimifolia*	1	0.185	44.13	62.2	18.3	19.5	0.102	1.448	14.229	
5300	R	*B. halimifolia*	0	0.3201	43.3	57.6	18.3	24.1	0.103	1.755	17.083	
5325	T	*B. halimifolia*	1	0.123	27.52	14	40.6	45.4	0.148	1.577	10.645	
5325	R	*B. halimifolia*	0	0.18	35.6	66.1	12	21.9	0.078	1.433	18.342	
5350	T	*B. halimifolia*	1	0.095	55.94	57.9	16.6	25.5	0.093	1.380	14.871	
5350	R	*B. halimifolia*	0	0.1334	33.62	62.7	16.6	20.7	0.092	1.584	17.249	
5375	R	*B. halimifolia*	0	0.458	43.7	63.9	14.6	21.5	0.078	1.493	19.139	
5400	R	*B. halimifolia*	0	0.176	28.58	67.5	17	15.5	0.106	1.381	12.969	

Appendix A. Raw data from observational study, continued

5400	T	*B. halimifolia*	1	0.2411	55.94	72.1	6.6	21.3	0.118	1.616	13.745
5425	R	*B. halimifolia*	0	0.1485	58.85	56.7	16.9	26.4	0.095	1.270	13.383
5450	R	*B. halimifolia*	0	0.1672	63.7	65.4	20.3	14.3	0.129	1.665	12.887
5450	T	*B. halimifolia*	1	0.2306	70.56	52.1	18.6	29.3	0.156	2.496	15.949
5475	R	*B. halimifolia*	0	0.166	70.8	6	35.4	58.6	0.092	1.197	13.039
5500	R	*B. halimifolia*	0	0.3688	60.92	52.4	16	31.6	0.133	1.420	10.686
5525	R	*B. halimifolia*	0	0.3747	38.75	61.3	10.6	28.1	0.106	1.742	16.488
5550	T	*B. halimifolia*	1	0.165	57.9	61.9	15.7	22.4	0.107	1.297	12.135
5550	R	*B. halimifolia*	0	0.1422	46.2	57.9	16.6	25.5	0.101	1.483	14.649
5575	T	*B. halimifolia*	1	0.0555	38.49	67.9	12.6	19.5	0.148	1.781	12.018
5575	R	*B. halimifolia*	0	0.15	54.43	65.5	14.6	19.9	0.115	1.678	14.581
5600	R	*B. halimifolia*	0	0.1679	58.72	75.5	8	16.5	0.130	2.139	16.404
5600	T	*B. halimifolia*	1	0.1505	52.9	77.5	4.6	17.9	0.121	2.008	16.597
5625	R	*B. halimifolia*	0	0.2068	56.13	63.9	10.6	25.5	0.143	1.713	11.971
5650	R	*B. halimifolia*	0	0.2255	66.55	65.9	14.3	19.8	0.102	1.703	16.733
5650	T	*B. halimifolia*	1	0.181	70.93	58.1	18	23.9	0.101	2.004	19.835
5675	R	*B. halimifolia*	0	0.129	66.96	65.3	12.3	22.4	0.101	1.391	13.779
5700	R	*B. halimifolia*	0	0.2276	65.18	60.7	16.3	23	0.106	1.293	12.170
5725	R	*B. halimifolia*	0	0.21	56.41	60.1	14	25.9	0.100	1.646	16.533
5725	T	*B. halimifolia*	1	0.3031	66	45	26.6	28.4	0.101	1.798	17.854
5750	T	*B. halimifolia*	1	0.2605	53.32	49	18.3	32.7	0.101	1.152	11.389
5750	R	*B. halimifolia*	0	0.285	63.33	61.5	15.2	23.3	0.093	1.456	15.726
5775	R	*B. halimifolia*	0	0.2423	44.92	64.7	12.3	23	0.106	1.565	14.828
5775	T	*B. halimifolia*	1	0.3122	39.65	61.4	18	20.6	0.073	1.110	15.200
5800	R	*B. halimifolia*	0	0.4027	24.3	63.3	16.6	20.1	0.091	1.594	17.593
5825	R	*B. halimifolia*	0	0.51	33.81	64.4	14.3	21.3	0.100	1.709	17.137
5825	T	*B. halimifolia*	1	0.4344	25.69	57.9	20	22.1	0.110	2.197	19.943
5850	T	*B. halimifolia*	1	0.616	51.09	67.5	14.6	17.9	0.294	5.257	17.901
5850	R	*B. halimifolia*	0	1.022	52.6	6.4	55	38.6	0.247	4.514	18.267
5875	T	*B. halimifolia*	1	0.286	42.68	60.1	16.6	23.3	0.090	1.577	17.555
5875	R	*B. halimifolia*	0	0.4	43.63	65.9	17	17.1	0.155	3.460	22.365
5900	T	*B. halimifolia*	1	0.526	45.08	63.5	12.6	23.9	0.117	2.208	18.911
5900	R	*B. halimifolia*	0	0.4885	37.75	52.7	22.6	24.7	0.085	1.918	22.645
5925	R	*B. halimifolia*	0	0.488	43.77	54.7	22.3	23	0.302	5.779	19.113
5950	R	*B. halimifolia*	0	0.383	36.58	65.9	19.7	14.4	0.211	3.764	17.875
5975	R	*B. halimifolia*	0	0.95	58.57	42.7	22.6	34.7	0.237	4.621	19.496

Appendix A. Raw data from observational study, continued

6000	T	*B. halimifolia*	1	0.2958	44.06	66.1	10.3	23.6	0.089	0.913	10.237
6000	R	*B. halimifolia*	0	0.2055	43.54	63.3	23.5	13.2	0.164	3.119	18.976
6025	R	*B. halimifolia*	0	0.166	44.25	51	22.3	26.7	0.196	3.024	15.453
6050	R	*B. halimifolia*	0	0.2883	54.92	50.7	20.9	28.4	0.186	2.846	15.286
6075	R	*B. halimifolia*	0	0.2682	58.48	63.3	16.9	19.8	0.171	2.937	17.131
6100	R	*B. halimifolia*	0	0.25	57.08	28.6	35.3	36.1	0.141	2.109	14.969
6100	T	*B. halimifolia*	1	0.2385	57.49	56.1	16.9	27	0.146	2.543	17.424
6125	R	*B. halimifolia*	0	0.3457	52.6	40.2	28.3	31.5	0.148	2.560	17.273
6150	R	*B. halimifolia*	0	0.48	38.27	51.9	24.6	23.5	0.064	0.871	13.596
6150	T	*B. halimifolia*	1	0.358	35.41	50.1	27.2	22.7	0.093	1.549	16.713
6175	R	*B. halimifolia*	0	0.287	23.89	56.1	16.9	27	0.132	2.041	15.442
6200	R	*B. halimifolia*	0	0.1182	47.55	37.9	32.6	29.5	0.091	1.160	12.728
6225	R	*B. halimifolia*	0	0.238	37.33	42.4	35.5	22.1	0.072	0.962	13.401
6250	R	*B. halimifolia*	0	0.2337	35.92	50.7	26.6	22.7	0.088	1.361	15.546
6250	T	*B. halimifolia*	1	0.3477	35.6	56.1	24.6	19.3	0.084	1.479	17.526
6275	R	*B. halimifolia*	0	0.2557	34.76	63.8	22.3	13.9	0.121	1.803	14.960
6300	R	*B. halimifolia*	0	0.158	32.99	61.3	20	18.7	0.095	1.068	11.232
6325	R	*B. halimifolia*	0	0.091	42.5	64.7	12.6	22.7	0.147	2.045	13.955
6350	R	*B. halimifolia*	0	0.1576	49.81	55.8	14.9	29.3	0.178	2.376	13.379
6350	T	*B. halimifolia*	1	0.201	44.52	65.7	12.3	22	0.102	1.530	14.963
6375	R	*B. halimifolia*	0	0.2454	49.58	31.3	36.3	32.4	0.159	2.341	14.765
6400	R	*B. halimifolia*	0	0.1919	51.6	15.3	48	36.7	0.185	2.569	13.887
6425	R	*B. halimifolia*	0	0.1511	50.67	39.5	26.6	33.9	0.147	2.009	13.661
6450	R	*B. halimifolia*	0	0.3422	44.39	41	28.3	30.7	0.162	2.440	15.055
6475	R	*B. halimifolia*	0	0.2442	34.34	50.1	26	23.9	0.152	2.267	14.933
6500	R	*B. halimifolia*	0	0.0902	33.79	50.7	18	31.3	0.128	2.176	16.996
6525	R	*B. halimifolia*	0	0.277	22.51	53.5	19.2	27.3	0.122	2.243	18.387
6550	R	*B. halimifolia*	0	0.1639	24.73	53	20.3	26.7	0.117	2.069	17.749
6575	R	*B. halimifolia*	0	0.2563	32.71	63.6	18.3	18.1	0.080	1.066	13.393
6600	R	*B. halimifolia*	0	0.165	27.81	75.3	13.2	11.5	0.084	0.989	11.721
6625	R	*B. halimifolia*	0	0.3356	19.7	67.9	22	10.1	0.103	1.512	14.713
6650	R	*B. halimifolia*	0	0.181	22.84	19	51	30	0.205	3.380	16.522
6675	R	*B. halimifolia*	0	0.1014	32.82	43.6	20	36.4	0.093	1.486	15.936
6700	R	*B. halimifolia*	0	0.1347	35.39	62.4	14.9	22.7	0.104	0.940	9.067
6725	R	*B. halimifolia*	0	0.1184	35.37	77	4.3	18.7	0.127	2.274	17.859
6750	R	*B. halimifolia*	0	0.1147	39.63	65.6	12.3	22.1	0.157	1.890	12.027

Appendix A. Raw data from observational study, continued

6775	R	*B. halimifolia*	0	0.22	43.78	74.7	14.9	10.4	0.064	0.734	11.390
6775	T	*B. halimifolia*	1	0.1991	42.9	70.7	10.6	18.7	0.062	0.936	15.120
6800	R	*B. halimifolia*	0	0.2425	40.82	56.7	16.3	27	0.071	1.034	14.602
6825	R	*B. halimifolia*	0	0.1897	44.68	54.1	16.9	29	0.081	1.267	15.628
6850	R	*B. halimifolia*	0	0.1405	40.71	49.4	16.9	33.7	0.091	1.280	13.989
6875	R	*B. halimifolia*	0	0.129	46.23	56.2	20	23.8	0.176	2.872	16.319
6900	R	*B. halimifolia*	0	.	49.35	.	.	.	.	.	.
6925	R	*B. halimifolia*	0	0.2054	30.22	14.4	58.4	27.2	0.180	3.714	20.666
6950	R	*B. halimifolia*	0	0.1992	40.49	70.1	7.2	22.7	0.181	2.198	12.143
6975	R	*B. halimifolia*	0	0.133	55.2	57.5	9.2	33.3	0.195	2.527	12.939
7000	R	*B. halimifolia*	0	0.3326	49.85	74.1	8.3	17.6	0.200	4.356	21.825
7025	R	*B. halimifolia*	0	0.1636	56.56	56.4	18.9	24.7	0.162	3.303	20.339
7050	R	*B. halimifolia*	0	0.1195	38.45	71.9	11.7	16.4	0.113	2.075	18.369
7075	R	*B. halimifolia*	0	0.1926	44.32	64.4	8.9	26.7	0.172	3.139	18.224
7100	R	*B. halimifolia*	0	0.1135	33.72	47	18.3	34.7	0.165	3.255	19.690
7125	R	*B. halimifolia*	0	0.1994	47.04	53.3	16.6	30.1	0.184	2.879	15.639
7150	R	*B. halimifolia*	0	0.1675	36.65	55	18.9	26.1	0.096	1.699	17.763
7175	R	*B. halimifolia*	0	0.1323	36.82	42.2	24.6	33.2	0.153	2.044	13.343
7200	R	*B. halimifolia*	0	0.11	47.93	41	28.6	30.4	0.156	2.471	15.850
7225	R	*B. halimifolia*	0	0.088	55.98	39	29.6	31.4	0.158	2.289	14.524
7250	R	*B. halimifolia*	0	0.1118	50.11	37	32.3	30.7	0.139	2.051	14.719
7275	R	*B. halimifolia*	0	0.1181	37.5	36.1	30.6	33.3	0.169	2.232	13.223
7300	R	*B. halimifolia*	0	0.115	21.99	59.7	19.9	20.4	0.133	2.063	15.558
8200	R	*B. halimifolia*	0	0.1064	21.69	61	22.3	16.7	0.060	1.083	18.098
8225	R	*B. halimifolia*	0	0.1306	14.91	57	20.6	22.4	0.053	1.129	21.213
8250	R	*B. halimifolia*	0	0.0823	14.04	70.1	8.3	21.6	0.093	1.500	16.089
8275	R	*B. halimifolia*	0	0.0496	36.55	74.1	12	13.9	0.060	0.826	13.771
8300	R	*B. halimifolia*	0	0.0504	52.86	74.4	11.5	14.1	0.098	1.482	15.092
8325	R	*B. halimifolia*	0	0.0482	58.2	63.6	24.3	12.1	0.070	1.000	14.370
8350	R	*B. halimifolia*	0	0.0511	67.19	69.9	11.7	18.4	0.097	1.468	15.198
8375	R	*B. halimifolia*	0	0.0681	52.13	47.6	26	26.4	0.133	1.906	14.334
8400	R	*B. halimifolia*	0	0.0541	45.15	60.4	13.7	25.9	0.118	1.787	15.197
8425	R	*B. halimifolia*	0	0.0884	53.89	64.7	12.3	23	0.190	2.786	14.687
8450	R	*B. halimifolia*	0	0.0877	43.38	65.3	20.3	14.4	0.133	1.906	14.334
8475	R	*B. halimifolia*	0	0.0404	42.9	73.3	18	8.7	0.048	0.683	14.099
8500	R	*B. halimifolia*	0	0.1688	20.34	69.6	14.3	16.1	0.146	0.516	17.207

Appendix A. Raw data from observational study, continued

8525	R	*B. halimifolia*	0	0.072	30.61	82.7	5.7	11.6	0.114	1.862	16.287
8550	R	*B. halimifolia*	0	0.1347	28.52	70.1	8.6	21.3	0.222	4.004	18.053
8575	R	*B. halimifolia*	0	0.1697	27.74	63.3	18	18.7	0.149	3.519	23.540
8600	R	*B. halimifolia*	0	0.1765	32.6	64.1	14.6	21.3	0.313	5.385	17.226
8625	R	*B. halimifolia*	0	0.1483	26.19	63.3	12	24.7	0.200	3.562	17.821
8650	R	*B. halimifolia*	0	0.1605	20.07	76.1	8.3	15.6	0.222	4.231	19.099
8675	R	*B. halimifolia*	0	0.223	19.62	47	35.8	17.2	0.299	5.128	17.151
8700	R	*B. halimifolia*	0	0.0721	18.48	79.5	6.8	13.7	0.206	3.890	18.925
8725	R	*B. halimifolia*	0	0.303	72.66	69.7	2.4	27.9	0.370	6.395	17.280
8750	R	*B. halimifolia*	0	0.2182	72.44	56.1	12.6	31.3	0.222	3.859	17.397
8775	R	*B. halimifolia*	0	0.19	14.74	7	46.4	46.6	0.438	8.230	18.800
8800	R	*B. halimifolia*	0	0.1638	34.75	26.3	20.9	52.8	0.265	4.008	15.152
8825	R	*B. halimifolia*	0	0.115	36.6	51.3	18	30.7	0.139	2.599	18.730
8850	R	*B. halimifolia*	0	0.15	37.63	41	22.6	36.4	0.292	4.033	13.816
8875	R	*B. halimifolia*	0	0.1173	23.32	55	22	23	0.183	2.993	16.392
8900	R	*B. halimifolia*	0	0.1055	24.93	67	18	15	0.139	2.562	18.431
8925	R	*B. halimifolia*	0	0.088	35.48	79.2	8.3	12.5	0.108	3.293	30.380
8950	R	*B. halimifolia*	0	0.0844	21.03	55.9	20.3	23.8	0.172	4.089	23.827
8975	R	*B. halimifolia*	0	0.1359	22.16	50.9	26.4	22.7	0.285	5.363	18.828
9000	R	*B. halimifolia*	0	0.0903	23.51	48.6	24.6	26.8	0.102	2.218	21.688
9025	R	*B. halimifolia*	0	0.111	21.47	58.5	18.3	23.2	0.298	6.009	20.186
9050	R	*B. halimifolia*	0	0.0855	13.13	61.3	8.6	30.1	0.145	2.958	20.410
9075	R	*B. halimifolia*	0	0.0732	25.52	81.3	6.3	12.4	0.181	3.206	17.689
9100	R	*B. halimifolia*	0	0.2022	9.16	69.2	11.2	19.6	0.251	4.753	18.925
9125	R	*B. halimifolia*	0	0.0959	9.69	52.7	18.6	28.7	0.170	2.880	16.910
9150	R	*B. halimifolia*	0	0.0493	52.46	75.5	13.2	11.3	0.107	1.870	17.411
9175	R	*B. halimifolia*	0	0.0435	43.66	65.9	18.3	15.8	0.084	1.313	15.631
9200	R	*B. halimifolia*	0	0.0299	49.2	71.8	14.9	13.3	0.087	1.453	16.735
9225	R	*B. halimifolia*	0	0.0718	39.67	61	16.3	22.7	0.141	1.750	12.434
9250	R	*B. halimifolia*	0	0.0756	28.23	67.2	9.2	23.6	0.150	2.980	19.823
9275	R	*B. halimifolia*	0	0.1012	31.31	61	16.3	22.7	0.150	4.022	26.825
9300	R	*B. halimifolia*	0	0.108	13.04	62.7	14.6	22.7	0.141	2.344	16.650
9325	R	*B. halimifolia*	0	0.1581	14.16	55.6	20.3	24.1	0.071	1.044	14.667
9350	R	*B. halimifolia*	0	0.0655	24.41	62.6	19.3	18.1	0.171	2.599	15.168
9375	R	*B. halimifolia*	0	0.0825	16.24	58.4	28.3	13.3	0.075	1.145	15.196
9400	R	*B. halimifolia*	0	0.0952	10.56	62.1	6.6	31.3	0.123	2.198	17.842

Appendix A. Raw data from observational study, continued

9425	R	*B. halimifolia*	0	0.1595	9.14	71.3	16	12.7	0.073	1.042	14.313	
9450	R	*B. halimifolia*	0	0.0913	28.23	78.1	8.9	13	0.135	2.179	16.087	
9475	R	*B. halimifolia*	0	0.074	31.31	65.3	14.3	20.4	0.200	2.090	10.438	
9500	R	*B. halimifolia*	0	0.135	12.25	79.6	12	8.4	0.066	1.559	23.722	
9525	R	*B. halimifolia*	0	0.1056	16.13	68.5	10.3	21.2	0.257	4.048	15.755	
9550	R	*B. halimifolia*	0	0.086	16.58	60.9	19.3	19.8	0.260	4.991	19.161	
9575	R	*B. halimifolia*	0	0.1358	24.26	47	31.6	21.4	0.326	5.628	17.243	
9600	R	*B. halimifolia*	0	0.084	27.25	69.2	13.2	17.6	0.208	3.947	18.985	
9625	R	*B. halimifolia*	0	0.0983	15.26	63.6	16.6	19.8	0.178	3.883	21.844	
9650	R	*B. halimifolia*	0	0.0962	12.04	57.3	16.3	26.4	0.201	4.176	20.803	
9675	R	*B. halimifolia*	0	0.125	8.94	65.5	13.7	20.8	0.378	6.589	17.428	
9700	R	*B. halimifolia*	0	0.12	18.15	11	58.4	30.6	0.366	6.633	18.112	
9725	R	*B. halimifolia*	0	0.0845	10.87	61.3	16.6	22.1	0.222	3.743	16.830	
9750	R	*B. halimifolia*	0	0.0829	11.87	43.6	22.3	34.1	0.259	4.329	16.702	
9775	R	*B. halimifolia*	0	0.082	22.14	48.2	20.3	31.5	0.192	3.044	15.848	
9800	R	*B. halimifolia*	0	0.1081	29.32	60.7	12	27.3	0.228	3.361	14.770	
9825	R	*B. halimifolia*	0	0.1041	30.47	48.1	20	31.9	0.189	3.015	15.958	
9850	R	*B. halimifolia*	0	0.0845	29.94	66.1	12.8	21.1	0.206	3.814	18.526	
9875	R	*B. halimifolia*	0	0.1145	21.32	29	26.3	44.7	0.319	5.466	17.130	
9900	R	*B. halimifolia*	0	0.0981	26.04	12.5	32.3	55.2	0.232	4.045	17.429	
9925	R	*B. halimifolia*	0	0.19	18.57	49.6	16.3	34.1	0.392	6.610	16.879	
9950	R	*B. halimifolia*	0	0.0988	15.88	38.5	25.3	36.2	0.214	4.588	21.438	
9975	R	*B. halimifolia*	0	0.0953	17.93	44.5	32.5	23	0.225	3.610	16.021	
10000	R	*B. halimifolia*	0	0.06	52.3	36.5	28.3	35.2	0.100	1.828	18.267	
10025	R	*B. halimifolia*	0	0.0305	28.24	46.7	20.3	33	0.211	3.027	14.382	
10075	R	*B. halimifolia*	0	0.04	33.01	53.1	20.6	26.3	0.247	5.176	20.971	
10100	R	*B. halimifolia*	0	0.0528	27.8	17.2	50.4	32.4	0.289	3.897	13.495	
10125	R	*B. halimifolia*	0	0.0207	21.43	52.38	18.42	29.2	0.089	1.343	15.163	
10150	R	*B. halimifolia*	0	0.0553	21.61	29.8	46.4	23.8	0.417	7.319	17.544	
10175	R	*B. halimifolia*	0	0.0645	21.96	65.6	18	16.4	0.243	3.827	15.767	
10200	R	*B. halimifolia*	0	0.0403	28.84	43	26.3	30.7	0.227	3.117	13.733	
10225	R	*B. halimifolia*	0	0.0522	32.16	48.9	25.3	25.8	0.170	2.776	16.304	
10250	R	*B. halimifolia*	0	0.036	30.71	47.6	26.3	26.1	0.213	3.222	15.105	
10275	R	*B. halimifolia*	0	0.0313	30.26	32.5	30.3	37.2	0.151	2.570	16.968	
10300	R	*B. halimifolia*	0	0.0433	39.19	52.4	20.6	27	0.211	3.061	14.472	
10325	R	*B. halimifolia*	0	0.045	34.91	34.7	30.6	34.7	0.223	2.589	11.625	

Appendix A. Raw data from observational study, continued

10350	R	*B. halimifolia*	0	0.0287	52.01	42.5	26.3	31.2	0.117	1.486	12.669	
10375	R	*B. halimifolia*	0	0.066	50.84	53.3	25.7	21	0.189	2.834	14.966	
10400	R	*B. halimifolia*	0	0.0785	48.61	36.2	34.3	29.5	0.341	5.620	16.470	
10425	R	*B. halimifolia*	0	0.073	67.26	43.4	32	24.6	0.139	1.909	13.765	
10450	R	*B. halimifolia*	0	0.0466	73.87	50.8	22.72	26.48	0.086	0.962	11.244	
10475	R	*B. halimifolia*	0	0.0455	57.57	42.8	30.6	26.6	0.099	1.228	12.465	
10500	R	*B. halimifolia*	0	0.0411	54.45	45.3	30.3	24.4	0.093	1.269	13.571	
10525	R	*B. halimifolia*	0	0.0485	72.95	62.2	21.4	16.4	0.099	1.362	13.733	
10550	R	*B. halimifolia*	0	0.044	61.7	57.6	23.7	18.7	0.097	1.259	12.939	
10575	R	*B. halimifolia*	0	0.0346	78.04	56.1	24	19.9	0.110	1.476	13.356	
10600	R	*B. halimifolia*	0	0.0446	82.43	36.5	34.3	29.2	0.126	1.540	12.237	
10625	R	*B. halimifolia*	0	0.0493	71.82	71.5	16	12.5	0.130	1.871	14.358	
10650	R	*B. halimifolia*	0	0.024	82.84	70.7	6	23.3	0.146	1.476	10.121	
10675	R	*B. halimifolia*	0	0.0258	66.14	74.7	4	21.3	0.102	1.385	13.600	
10700	R	*B. halimifolia*	0	0.031	76.02	58.2	16.6	25.2	0.092	1.142	12.463	
10725	R	*B. halimifolia*	0	0.0228	66.72	38.8	37.3	23.9	0.150	0.161	14.384	
10750	R	*B. halimifolia*	0	0.0344	62.6	75.5	7.7	16.8	0.129	1.839	14.299	
10775	R	*B. halimifolia*	0	0.023	59.38	30.8	30.6	38.6	0.160	2.164	13.525	
10800	R	*B. halimifolia*	0	0.0436	75.6	34.7	30.3	35	0.219	3.114	14.206	
10825	R	*B. halimifolia*	0	0.0291	82.53	43.2	26.3	30.5	0.185	2.380	12.843	
10850	R	*B. halimifolia*	0	0.039	78.25	26.1	37.3	36.6	0.141	1.808	12.855	
10875	R	*B. halimifolia*	0	0.0392	77.51	36.8	30.6	32.6	0.218	2.423	11.116	
10900	R	*B. halimifolia*	0	0.04	83.19	35	34.6	30.4	0.285	3.128	10.972	
10925	R	*B. halimifolia*	0	0.0394	87.64	62.5	18	19.5	0.113	1.812	16.084	
10950	R	*B. halimifolia*	0	0.037	88.3	76.1	10	13.9	0.104	1.329	12.814	
10975	R	*B. halimifolia*	0	0.0307	87.67	61.3	18.3	20.4	0.124	1.369	11.056	
11000	R	*B. halimifolia*	0	0.029	87.68	50.1	24.3	25.6	0.155	2.323	15.020	
11025	R	*B. halimifolia*	0	0.0232	86.4	83.3	10.3	6.4	0.040	0.511	12.761	
11050	R	*B. halimifolia*	0	0.0262	85.57	82.2	10	7.8	0.115	1.760	15.250	
11075	R	*B. halimifolia*	0	0.019	85.62	73.3	4.6	22.1	0.061	0.503	8.178	
11100	R	*B. halimifolia*	0	0.0227	87.24	83.2	11.2	5.6	0.069	0.581	8.399	
11125	R	*B. halimifolia*	0	0.0306	85.4	44.5	28.9	26.6	0.147	2.111	14.316	
11150	R	*B. halimifolia*	0	0.03	84.44	51.3	29.3	19.4	0.189	3.501	18.529	
11175	R	*B. halimifolia*	0	0.0256	85.92	61.5	18.9	19.6	0.144	1.762	12.241	
11200	R	*B. halimifolia*	0	0.0295	85.75	48.4	28.9	22.7	0.182	2.480	13.643	
11225	R	*B. halimifolia*	0	0.06	81.15	55.3	26.3	18.4	0.158	1.960	12.395	

Appendix A. Raw data from observational study, continued

11250	R	*B. halimifolia*	0	0.0716	75.79	23.1	40.6	36.3	0.197	2.841	14.446
11275	R	*B. halimifolia*	0	0.1037	74.05	65.8	14.9	19.3	0.147	2.854	19.361
11300	R	*B. halimifolia*	0	0.0814	57.24	33.5	34.3	32.2	0.320	4.026	12.584
0	R	*I. vomitoria*	0	6.66	98.15	73.6	10.6	15.8	0.201	3.136	15.616
100	R	*I. vomitoria*	0	7.53	98.15	30.4	43	26.6	0.386	4.648	12.035
200	R	*I. vomitoria*	0	9.4	98.15	61.8	18.3	19.9	0.373	5.527	14.819
300	R	*I. vomitoria*	0	9.03	98.15	59.1	30.9	10	0.459	5.935	12.939
400	R	*I. vomitoria*	0	7.4	98.15	13.8	43.6	42.6	0.291	4.170	14.353
500	R	*I. vomitoria*	0	9.28	98.15	56.8	16.9	26.3	0.173	2.682	15.472
600	R	*I. vomitoria*	0	8.49	98.15	57.1	18.3	24.6	0.324	4.831	14.893
700	R	*I. vomitoria*	0	3.567	98.15	67	14	19	0.047	0.424	8.967
800	R	*I. vomitoria*	0	8.59	98.15	34.4	35	30.6	0.253	4.045	15.992
900	R	*I. vomitoria*	0	10.88	98.15	35	38.4	26.6	0.318	4.948	15.583
1000	R	*I. vomitoria*	0	7.38	98.15	69.5	18.6	11.9	0.280	4.108	14.654
1100	R	*I. vomitoria*	0	10.1	98.15	27	39	34	0.289	3.897	13.495
1200	R	*I. vomitoria*	0	11.53	98.15	.	.	.	0.451	6.415	14.211
1300	R	*I. vomitoria*	0	8.94	98.15	74.4	6.4	19.2	0.467	6.793	14.551
1400	R	*I. vomitoria*	0	6.43	98.15	59.5	24.3	16.2	0.215	2.749	12.772
1500	R	*I. vomitoria*	0	10	98.15	37	52	11	0.436	5.976	13.693
1600	R	*I. vomitoria*	0	8.34	98.15	73.2	12.3	14.5	0.411	6.091	14.814
1700	R	*I. vomitoria*	0	7.3	98.15	30.4	51	18.6	0.272	3.751	13.791
1800	R	*I. vomitoria*	0	9	98.15	29	49.7	21.3	0.317	4.648	14.678
1900	R	*I. vomitoria*	0	8	98.15	43	37	20	0.282	4.293	15.245
2000	R	*I. vomitoria*	0	11.7	98.15	32.4	27.2	40.4	0.381	5.262	13.821
2100	R	*I. vomitoria*	0	8.75	98.15	67.6	16.3	16.1	0.154	2.002	12.975
2200	R	*I. vomitoria*	0	11.63	98.15	13.8	26.4	59.8	0.284	3.950	13.903
2300	R	*I. vomitoria*	0	5.7	98.15	39.4	36.3	24.3	0.095	1.060	11.213
2400	R	*I. vomitoria*	0	8.725	98.15	67.1	10.3	22.6	0.375	6.057	16.170
2500	R	*I. vomitoria*	0	10.28	98.15	65.1	15.4	19.5	0.312	5.125	16.407
2600	R	*I. vomitoria*	0	9.76	98.15	59.5	21.2	19.3	0.324	4.831	14.893
2700	R	*I. vomitoria*	0	9.66	98.15	70.9	13.2	15.9	0.386	5.829	15.113
2800	R	*I. vomitoria*	0	10.5	98.15	67.2	11.5	21.3	0.345	5.384	15.610
2900	R	*I. vomitoria*	0	5.76	98.15	59.5	18.9	21.6	0.217	2.588	11.923
3000	R	*I. vomitoria*	0	9.19	98.15	39.6	39	21.4	0.389	5.392	13.861
3100	R	*I. vomitoria*	0	7.91	98.15	64.1	22	13.9	0.364	5.916	16.244
3200	R	*I. vomitoria*	0	8.5	98.15	57.5	20.6	21.9	0.263	3.985	15.147

Appendix A. Raw data from observational study, continued

3300	R	*I. vomitoria*	0	9.3	98.15	45.8	19.6	34.6	0.281	4.065	14.476	
3400	R	*I. vomitoria*	0	7.11	98.15	60.5	31.2	8.3	0.433	6.931	15.991	
3500	R	*I. vomitoria*	0	7.1	98.15	64.1	18.6	17.3	0.446	6.784	15.200	
3600	R	*I. vomitoria*	0	9.66	98.15	49.1	40.6	10.3	0.472	6.799	14.396	
3700	R	*I. vomitoria*	0	8.72	98.15	39	33.2	27.8	0.208	2.956	14.241	
3800	R	*I. vomitoria*	0	13	98.15	67.5	11.2	21.3	0.418	7.770	18.566	
3900	R	*I. vomitoria*	0	15.43	98.15	48.9	39.8	11.3	0.464	9.034	19.483	
4000	R	*I. vomitoria*	0	14.6	98.15	59.5	25.5	15	1.079	11.161	10.342	
4100	R	*I. vomitoria*	0	18.46	98.15	56.8	29.9	13.3	0.482	8.177	16.957	
4200	R	*I. vomitoria*	0	16.25	97.87	68.8	19.3	11.9	0.420	9.002	21.415	
4300	R	*I. vomitoria*	0	6.07	97.69	59.9	18.9	21.2	0.151	3.122	20.669	
4400	R	*I. vomitoria*	0	3.93	90.81	51.6	28	20.4	0.079	1.297	16.380	
4500	R	*I. vomitoria*	0	3.9	89.67	39.2	42.7	18.1	0.060	0.240	4.003	
4600	R	*I. vomitoria*	0	4.72	88.53	37	35.2	27.8	0.148	1.914	12.927	
4625	R	*I. vomitoria*	0	3.891	85.58	52.2	22.3	25.5	0.071	1.055	14.817	
4650	R	*I. vomitoria*	0	0.892	73.73	57.3	26.3	16.4	0.068	0.899	13.266	
4650	T	*I. vomitoria*	1	1.615	74.61	62.2	13.4	24.4	0.063	0.904	14.407	
4675	R	*I. vomitoria*	0	0.4494	60.46	81.5	7.2	11.3	0.068	1.037	15.159	
4700	R	*I. vomitoria*	0	0.463	59.45	63.3	14.3	22.4	0.091	1.229	13.512	
4725	R	*I. vomitoria*	0	0.2	58.7	57.9	24.6	17.5	0.106	0.957	9.032	
4725	T	*I. vomitoria*	1	0.2649	54.47	74.7	14.2	11.1	0.068	1.099	16.203	
4750	R	*I. vomitoria*	0	0.3425	43.58	35.8	36	28.2	0.083	0.702	8.454	
4775	R	*I. vomitoria*	1	0.2601	40.31	59	20.3	20.7	0.149	2.775	18.569	
4800	R	*I. vomitoria*	0	0.3266	40.16	76.4	11.8	11.8	0.053	0.694	13.180	
4825	T	*I. vomitoria*	1	0.645	46.6	77	8	15	0.043	0.550	12.858	
4825	R	*I. vomitoria*	0	2.652	48.08	49	22	29	0.060	0.967	16.123	
4850	R	*I. vomitoria*	0	1.354	71.21	51	21.7	27.3	0.127	1.887	14.864	
4875	R	*I. vomitoria*	0	1.5774	79.52	54.4	28.5	17.1	0.114	1.741	15.319	
4900	R	*I. vomitoria*	0	3.735	85.12	10.2	68	21.8	0.181	3.492	19.271	
4925	R	*I. vomitoria*	0	4.35	84.05	64.7	17.8	17.5	0.379	6.877	18.142	
4950	R	*I. vomitoria*	0	4.75	81.5	77.3	13.5	9.2	0.556	8.788	15.808	
4975	R	*I. vomitoria*	0	2.28	83.92	52.7	19.7	27.6	0.159	3.044	19.125	
5000	R	*I. vomitoria*	0	1.012	84.23	79.5	3.2	17.3	0.153	2.412	15.736	
5025	R	*I. vomitoria*	0	2.025	85.44	59	20.9	20.1	0.103	1.951	18.928	
5050	R	*I. vomitoria*	0	1.4	79.42	56.1	20.6	23.3	0.174	2.207	12.677	
5075	R	*I. vomitoria*	0	1.444	71.6	53	20.9	26.1	0.104	1.477	14.220	

Appendix A. Raw data from observational study, continued

5100	R	*I. vomitoria*	0	0.4307	48.14	61.6	14.3	24.1	0.105	1.699	16.182
5125	R	*I. vomitoria*	0	0.332	48.52	59	14	27	0.123	1.952	15.829
5150	R	*I. vomitoria*	0	0.547	46.11	59.6	18.3	22.1	0.124	1.595	12.843
5175	T	*I. vomitoria*	1	0.523	24.82	69.5	17.2	13.3	0.093	1.220	13.162
5175	R	*I. vomitoria*	0	0.4485	42.44	68.4	16.6	15	0.076	1.050	13.791
5200	R	*I. vomitoria*	0	0.426	40.58	65	16.3	18.7	0.062	1.059	17.190
5225	R	*I. vomitoria*	0	1.601	70.58	45	18	37	0.244	3.834	15.703
5250	R	*I. vomitoria*	0	1.788	70.47	59.6	18.3	22.1	0.243	2.772	11.408
5275	R	*I. vomitoria*	0	0.2601	39.15	65	12.3	22.7	0.130	2.107	16.204
5275	T	*I. vomitoria*	1	0.4578	43.38	51	18.3	30.7	0.135	2.448	18.186
5300	R	*I. vomitoria*	0	0.3201	43.3	57.6	18.3	24.1	0.103	1.755	17.083
5325	R	*I. vomitoria*	0	0.18	35.6	66.1	12	21.9	0.078	1.433	18.342
5350	R	*I. vomitoria*	0	0.1334	33.62	62.7	16.6	20.7	0.092	1.584	17.249
5375	R	*I. vomitoria*	0	0.458	43.7	63.9	14.6	21.5	0.078	1.493	19.139
5400	R	*I. vomitoria*	0	0.176	28.58	67.5	17	15.5	0.106	1.381	12.969
5425	R	*I. vomitoria*	0	0.1485	58.85	56.7	16.9	26.4	0.095	1.270	13.383
5450	R	*I. vomitoria*	0	0.1672	63.7	65.4	20.3	14.3	0.129	1.665	12.887
5475	R	*I. vomitoria*	0	0.166	70.8	6	35.4	58.6	0.092	1.197	13.039
5500	R	*I. vomitoria*	0	0.3688	60.92	52.4	16	31.6	0.133	1.420	10.686
5525	R	*I. vomitoria*	0	0.3747	38.75	61.3	10.6	28.1	0.106	1.742	16.488
5550	R	*I. vomitoria*	0	0.1422	46.2	57.9	16.6	25.5	0.101	1.483	14.649
5575	R	*I. vomitoria*	0	0.15	54.43	65.5	14.6	19.9	0.115	1.678	14.581
5575	T	*I. vomitoria*	1	0.157	41.4	62.2	12.6	25.2	0.108	1.781	16.462
5600	T	*I. vomitoria*	1	0.349	51.47	57.1	20.3	22.6	0.142	1.908	13.438
5600	R	*I. vomitoria*	0	0.1679	58.72	75.5	8	16.5	0.130	2.139	16.404
5625	R	*I. vomitoria*	0	0.2068	56.13	63.9	10.6	25.5	0.143	1.713	11.971
5650	R	*I. vomitoria*	0	0.2255	66.55	65.9	14.3	19.8	0.102	1.703	16.733
5675	R	*I. vomitoria*	0	0.129	66.96	65.3	12.3	22.4	0.101	1.391	13.779
5700	R	*I. vomitoria*	0	0.2276	65.18	60.7	16.3	23	0.106	1.293	12.170
5725	R	*I. vomitoria*	0	0.21	56.41	60.1	14	25.9	0.100	1.646	16.533
5750	R	*I. vomitoria*	0	0.285	63.33	61.5	15.2	23.3	0.093	1.456	15.726
5775	T	*I. vomitoria*	1	0.59	34.89	67.6	16.3	16.1	0.115	1.595	13.921
5775	R	*I. vomitoria*	0	0.2423	44.92	64.7	12.3	23	0.106	1.565	14.828
5800	T	*I. vomitoria*	1	0.1873	23.64	51.6	26.3	22.1	0.087	1.475	17.007
5800	R	*I. vomitoria*	0	0.4027	24.3	63.3	16.6	20.1	0.091	1.594	17.593
5825	R	*I. vomitoria*	0	0.51	33.81	64.4	14.3	21.3	0.100	1.709	17.137

Appendix A. Raw data from observational study, continued

5825	T	*I. vomitoria*	1	0.3682	37.83	62.1	12.3	25.6	0.170	3.789	22.240	
5850	T	*I. vomitoria*	1	0.2751	50.32	61.5	13.2	25.3	0.199	3.512	17.680	
5850	R	*I. vomitoria*	0	1.022	52.6	6.4	55	38.6	0.247	4.514	18.267	
5875	T	*I. vomitoria*	1	0.2243	43.73	63.5	13.2	23.3	0.141	2.107	14.957	
5875	R	*I. vomitoria*	0	0.4	43.63	65.9	17	17.1	0.155	3.460	22.365	
5900	R	*I. vomitoria*	0	0.4885	37.75	52.7	22.6	24.7	0.085	1.918	22.645	
5900	T	*I. vomitoria*	1	0.4583	44.88	54.4	20	25.6	0.173	5.047	29.205	
5925	T	*I. vomitoria*	1	0.35	42.39	49.3	26.6	24.1	0.203	3.387	16.644	
5925	R	*I. vomitoria*	0	0.488	43.77	54.7	22.3	23	0.302	5.779	19.113	
5950	R	*I. vomitoria*	0	0.383	36.58	65.9	19.7	14.4	0.211	3.764	17.875	
5975	T	*I. vomitoria*	1	0.3235	39.87	52.5	18.3	29.2	0.190	3.543	18.669	
5975	R	*I. vomitoria*	0	0.95	58.57	42.7	22.6	34.7	0.237	4.621	19.496	
6000	T	*I. vomitoria*	1	0.535	40.77	62.9	19	18.1	0.101	1.128	11.210	
6000	R	*I. vomitoria*	0	0.2055	43.54	63.3	23.5	13.2	0.164	3.119	18.976	
6025	T	*I. vomitoria*	1	0.1942	49.03	53.6	24	22.4	0.131	1.914	14.575	
6025	R	*I. vomitoria*	0	0.166	44.25	51	22.3	26.7	0.196	3.024	15.453	
6050	R	*I. vomitoria*	0	0.2883	54.92	50.7	20.9	28.4	0.186	2.846	15.286	
6050	T	*I. vomitoria*	1	0.2865	45.22	53	24.9	22.1	0.186	2.846	15.286	
6075	R	*I. vomitoria*	0	0.2682	58.48	63.3	16.9	19.8	0.171	2.937	17.131	
6075	T	*I. vomitoria*	1	0.2815	58.97	43.3	26	30.7	0.169	2.997	17.691	
6100	R	*I. vomitoria*	0	0.25	57.08	28.6	35.3	36.1	0.141	2.109	14.969	
6100	T	*I. vomitoria*	1	0.1545	58.28	49.3	22	28.7	0.163	2.775	16.988	
6125	T	*I. vomitoria*	1	0.1809	46.73	50.1	22	27.9	0.188	2.805	14.882	
6125	R	*I. vomitoria*	0	0.3457	52.6	40.2	28.3	31.5	0.148	2.560	17.273	
6150	R	*I. vomitoria*	0	0.48	38.27	51.9	24.6	23.5	0.064	0.871	13.596	
6150	T	*I. vomitoria*	1	0.538	39.72	62.4	16.9	20.7	0.122	2.098	17.156	
6175	R	*I. vomitoria*	0	0.287	23.89	56.1	16.9	27	0.132	2.041	15.442	
6175	T	*I. vomitoria*	1	0.1934	40.04	47.9	28.6	23.5	0.120	2.414	20.108	
6200	R	*I. vomitoria*	0	0.1182	47.55	37.9	32.6	29.5	0.091	1.160	12.728	
6200	T	*I. vomitoria*	1	0.454	45.07	43.9	30	26.1	0.104	1.338	12.829	
6225	R	*I. vomitoria*	0	0.238	37.33	42.4	35.5	22.1	0.072	0.962	13.401	
6225	T	*I. vomitoria*	1	0.4316	39.66	47.8	28.9	23.3	0.094	1.529	16.195	
6250	R	*I. vomitoria*	0	0.2337	35.92	50.7	26.6	22.7	0.088	1.361	15.546	
6275	T	*I. vomitoria*	1	0.3735	34.74	65.9	17.7	16.4	0.070	0.912	12.973	
6275	R	*I. vomitoria*	0	0.2557	34.76	63.8	22.3	13.9	0.121	1.803	14.960	
6300	R	*I. vomitoria*	0	0.158	32.99	61.3	20	18.7	0.095	1.068	11.232	

Appendix A. Raw data from observational study, continued

6325	R	*I. vomitoria*	0	0.091	42.5	64.7	12.6	22.7	0.147	2.045	13.955
6325	T	*I. vomitoria*	1	0.245	26.85	75.1	2.9	22	0.093	1.332	14.375
6350	R	*I. vomitoria*	0	0.1576	49.81	55.8	14.9	29.3	0.178	2.376	13.379
6375	R	*I. vomitoria*	0	0.2454	49.58	31.3	36.3	32.4	0.159	2.341	14.765
6375	T	*I. vomitoria*	1	0.2771	43.84	35.2	45.8	19	0.067	1.305	19.372
6400	R	*I. vomitoria*	0	0.1919	51.6	15.3	48	36.7	0.185	2.569	13.887
6425	R	*I. vomitoria*	0	0.1511	50.67	39.5	26.6	33.9	0.147	2.009	13.661
6450	R	*I. vomitoria*	0	0.3422	44.39	41	28.3	30.7	0.162	2.440	15.055
6475	R	*I. vomitoria*	0	0.2442	34.34	50.1	26	23.9	0.152	2.267	14.933
6500	R	*I. vomitoria*	0	0.0902	33.79	50.7	18	31.3	0.128	2.176	16.996
6525	R	*I. vomitoria*	0	0.277	22.51	53.5	19.2	27.3	0.122	2.243	18.387
6525	T	*I. vomitoria*	1	0.23	35.54	55.9	20.3	23.8	0.127	2.349	18.460
6550	T	*I. vomitoria*	1	0.2865	34.1	67.9	18	14.1	0.067	1.052	15.611
6550	R	*I. vomitoria*	0	0.1639	24.73	53	20.3	26.7	0.117	2.069	17.749
6575	T	*I. vomitoria*	1	0.3034	36.45	64.1	20.6	15.3	0.062	0.801	13.025
6575	R	*I. vomitoria*	0	0.2563	32.71	63.6	18.3	18.1	0.080	1.066	13.393
6600	R	*I. vomitoria*	0	0.165	27.81	75.3	13.2	11.5	0.084	0.989	11.721
6625	T	*I. vomitoria*	1	0.148	34.24	76.1	10	13.9	0.051	0.671	13.068
6625	R	*I. vomitoria*	0	0.3356	19.7	67.9	22	10.1	0.103	1.512	14.713
6650	R	*I. vomitoria*	0	0.181	22.84	19	51	30	0.205	3.380	16.522
6650	T	*I. vomitoria*	1	0.3465	17.03	52.7	20.6	26.7	0.184	3.694	20.068
6675	R	*I. vomitoria*	0	0.1014	32.82	43.6	20	36.4	0.093	1.486	15.936
6700	R	*I. vomitoria*	0	0.1347	35.39	62.4	14.9	22.7	0.104	0.940	9.067
6700	T	*I. vomitoria*	1	0.1787	20.8	61.6	24	14.4	0.053	0.796	15.110
6725	R	*I. vomitoria*	0	0.1184	35.37	77	4.3	18.7	0.127	2.274	17.859
6750	T	*I. vomitoria*	1	0.237	16.61	47.6	26.3	26.1	0.200	1.989	9.924
6750	R	*I. vomitoria*	0	0.1147	39.63	65.6	12.3	22.1	0.157	1.890	12.027
6775	R	*I. vomitoria*	0	0.22	43.78	74.7	14.9	10.4	0.064	0.734	11.390
6800	R	*I. vomitoria*	0	0.2425	40.82	56.7	16.3	27	0.071	1.034	14.602
6825	R	*I. vomitoria*	0	0.1897	44.68	54.1	16.9	29	0.081	1.267	15.628
6850	R	*I. vomitoria*	0	0.1405	40.71	49.4	16.9	33.7	0.091	1.280	13.989
6875	R	*I. vomitoria*	0	0.129	46.23	56.2	20	23.8	0.176	2.872	16.319
6900	R	*I. vomitoria*	0	.	49.35	.	.	.	.	.	.
6925	R	*I. vomitoria*	0	0.2054	30.22	14.4	58.4	27.2	0.180	3.714	20.666
6950	R	*I. vomitoria*	0	0.1992	40.49	70.1	7.2	22.7	0.181	2.198	12.143
6975	R	*I. vomitoria*	0	0.133	55.2	57.5	9.2	33.3	0.195	2.527	12.939

Appendix A. Raw data from observational study, continued

7000	R	*I. vomitoria*	0	0.3326	49.85	74.1	8.3	17.6	0.200	4.356	21.825
7025	R	*I. vomitoria*	0	0.1636	56.56	56.4	18.9	24.7	0.162	0.303	20.339
7050	R	*I. vomitoria*	0	0.1195	38.45	71.9	11.7	16.4	0.113	2.075	18.369
7050	T	*I. vomitoria*	1	0.1787	55.82	75.6	10.3	14.1	0.113	2.075	18.369
7075	R	*I. vomitoria*	0	0.1926	44.32	64.4	8.9	26.7	0.172	3.139	18.224
7100	T	*I. vomitoria*	1	0.2004	35.21	42.7	24.6	32.7	0.248	4.342	17.523
7100	R	*I. vomitoria*	0	0.1135	33.72	47	18.3	34.7	0.165	3.255	19.690
7125	R	*I. vomitoria*	0	0.1994	47.04	53.3	16.6	30.1	0.184	2.879	15.639
7150	R	*I. vomitoria*	0	0.1675	36.65	55	18.9	26.1	0.096	1.699	17.763
7175	R	*I. vomitoria*	0	0.1323	36.82	42.2	24.6	33.2	0.153	2.044	13.343
7175	T	*I. vomitoria*	1	0.1677	31.6	41	24.6	34.4	0.169	2.626	15.541
7200	R	*I. vomitoria*	0	0.11	47.93	41	28.6	30.4	0.156	2.471	15.850
7225	R	*I. vomitoria*	0	0.088	55.98	39	29.6	31.4	0.158	2.289	14.524
7250	R	*I. vomitoria*	0	0.1118	50.11	37	32.3	30.7	0.139	2.051	14.719
7250	T	*I. vomitoria*	1	0.1204	47.01	35	30	35	0.135	2.071	15.361
7275	R	*I. vomitoria*	0	0.1181	37.5	36.1	30.6	33.3	0.169	2.232	13.223
7275	T	*I. vomitoria*	1	0.2245	29.15	35	30	35	0.142	2.452	17.257
7300	R	*I. vomitoria*	0	0.115	21.99	59.7	19.9	20.4	0.133	2.063	15.558
8200	T	*I. vomitoria*	1	0.181	38.16	55.3	23.7	21	0.080	1.164	14.533
8200	R	*I. vomitoria*	0	0.1064	21.69	61	22.3	16.7	0.060	1.083	18.098
8225	T	*I. vomitoria*	1	0.1115	15.47	64.7	12.3	23	0.060	0.946	15.876
8225	R	*I. vomitoria*	0	0.1306	14.91	57	20.6	22.4	0.053	0.129	21.213
8250	T	*I. vomitoria*	1	0.0621	21.34	61.3	16.3	22.4	0.036	0.322	8.943
8250	R	*I. vomitoria*	0	0.0823	14.04	70.1	8.3	21.6	0.093	1.500	16.089
8275	R	*I. vomitoria*	0	0.0496	36.55	74.1	12	13.9	0.060	0.826	13.771
8300	T	*I. vomitoria*	1	0.0514	53.12	75	10	15	0.040	0.459	11.424
8300	R	*I. vomitoria*	0	0.0504	52.86	74.4	11.5	14.1	0.098	1.482	15.092
8325	R	*I. vomitoria*	0	0.0482	58.2	63.6	24.3	12.1	0.070	1.000	14.370
8350	R	*I. vomitoria*	0	0.0511	67.19	69.9	11.7	18.4	0.097	1.468	15.198
8375	R	*I. vomitoria*	0	0.0681	52.13	47.6	26	26.4	0.133	1.906	14.334
8400	R	*I. vomitoria*	0	0.0541	45.15	60.4	13.7	25.9	0.118	1.787	15.197
8400	T	*I. vomitoria*	1	0.1178	33.06	58.1	24.6	17.3	0.175	2.762	15.739
8425	R	*I. vomitoria*	0	0.0884	53.89	64.7	12.3	23	0.190	2.786	14.687
8450	R	*I. vomitoria*	0	0.0877	43.38	65.3	20.3	14.4	0.133	1.906	14.334
8475	T	*I. vomitoria*	1	0.058	52.84	74.4	8.9	16.7	0.065	0.601	9.205
8475	R	*I. vomitoria*	0	0.0404	42.9	73.3	18	8.7	0.048	0.683	14.099

Appendix A. Raw data from observational study, continued

8500	R	*I. vomitoria*	0	0.1688	20.34	69.6	14.3	16.1	0.146	2.516	17.207
8525	R	*I. vomitoria*	0	0.072	30.61	82.7	5.7	11.6	0.114	1.862	16.287
8550	T	*I. vomitoria*	1	0.1415	28.5	27	55	18	0.322	5.514	17.110
8550	R	*I. vomitoria*	0	0.1347	28.52	70.1	8.6	21.3	0.222	4.004	18.053
8575	T	*I. vomitoria*	1	0.1078	30.45	62.1	16.3	21.6	0.232	4.583	19.788
8575	R	*I. vomitoria*	0	0.1697	27.74	63.3	18	18.7	0.149	3.519	23.540
8600	R	*I. vomitoria*	0	0.1765	32.6	64.1	14.6	21.3	0.313	5.385	17.226
8600	T	*I. vomitoria*	1	0.1508	32.95	63.9	22	14.1	0.198	4.429	22.406
8625	R	*I. vomitoria*	0	0.1483	26.19	63.3	12	24.7	0.200	3.562	17.821
8650	T	*I. vomitoria*	1	0.12	14.97	68.4	12.9	18.7	0.192	3.472	18.127
8650	R	*I. vomitoria*	0	0.1605	20.07	76.1	8.3	15.6	0.222	4.231	19.099
8675	R	*I. vomitoria*	0	0.223	19.62	47	35.8	17.2	0.299	5.128	17.151
8700	R	*I. vomitoria*	0	0.0721	18.48	79.5	6.8	13.7	0.206	3.890	18.925
8725	R	*I. vomitoria*	0	0.303	72.66	69.7	2.4	27.9	0.370	6.395	17.280
8750	R	*I. vomitoria*	0	0.2182	72.44	56.1	12.6	31.3	0.222	3.859	17.397
8775	R	*I. vomitoria*	0	0.19	14.74	7	46.4	46.6	0.438	8.230	18.800
8800	R	*I. vomitoria*	0	0.1638	34.75	26.3	20.9	52.8	0.265	4.008	15.152
8825	R	*I. vomitoria*	0	0.115	36.6	51.3	18	30.7	0.139	2.599	18.730
8850	R	*I. vomitoria*	0	0.15	37.63	41	22.6	36.4	0.292	4.033	13.816
8875	T	*I. vomitoria*	1	0.105	28.7	46.7	24.3	29	0.151	2.203	14.635
8875	R	*I. vomitoria*	0	0.1173	23.32	55	22	23	0.183	2.993	16.392
8900	R	*I. vomitoria*	0	0.1055	24.93	67	18	15	0.139	2.562	18.431
8925	R	*I. vomitoria*	0	0.088	35.48	79.2	8.3	12.5	0.168	3.093	18.368
8925	T	*I. vomitoria*	1	0.0809	26.66	73.8	10.6	15.6	0.123	2.486	20.254
8950	T	*I. vomitoria*	1	0.0656	26.23	75	12.9	12.1	0.147	3.131	21.232
8950	R	*I. vomitoria*	0	0.0844	21.03	55.9	20.3	23.8	0.172	4.089	23.827
8975	R	*I. vomitoria*	0	0.1359	22.16	50.9	26.4	22.7	0.285	5.363	18.828
9000	R	*I. vomitoria*	0	0.0903	23.51	48.6	24.6	26.8	0.102	2.218	21.688
9025	T	*I. vomitoria*	1	0.1031	17.02	58.7	16.3	25	0.123	2.360	19.183
9025	R	*I. vomitoria*	0	0.111	21.47	58.5	18.3	23.2	0.298	6.009	20.186
9050	R	*I. vomitoria*	0	0.0855	13.13	61.3	8.6	30.1	0.145	2.958	20.410
9075	R	*I. vomitoria*	0	0.0732	25.52	81.3	6.3	12.4	0.181	3.206	17.689
9100	R	*I. vomitoria*	0	0.2022	9.16	69.2	11.2	19.6	0.251	4.753	18.925
9125	R	*I. vomitoria*	0	0.0959	9.69	52.7	18.6	28.7	0.170	2.880	16.910
9125	T	*I. vomitoria*	1	0.1254	26.23	55	18	27	0.075	1.366	18.175
9150	R	*I. vomitoria*	0	0.0493	52.46	75.5	13.2	11.3	0.107	1.870	17.411

Appendix A. Raw data from observational study, continued

9150	T	*I. vomitoria*	1	0.086	50.04	57.6	20.6	21.8	0.079	1.407	17.736
9175	R	*I. vomitoria*	0	0.0435	43.66	65.9	18.3	15.8	0.084	1.313	15.631
9200	T	*I. vomitoria*	1	0.0354	58.02	62.2	16.6	21.2	0.143	1.498	10.471
9200	R	*I. vomitoria*	0	0.0299	49.2	71.8	14.9	13.3	0.087	1.453	16.735
9225	R	*I. vomitoria*	0	0.0718	39.67	61	16.3	22.7	0.141	1.750	12.434
9250	R	*I. vomitoria*	0	0.0756	28.23	67.2	9.2	23.6	0.150	2.980	19.823
9275	R	*I. vomitoria*	0	0.1012	31.31	61	16.3	22.7	0.150	4.022	26.825
9300	R	*I. vomitoria*	0	0.108	13.04	62.7	14.6	22.7	0.141	2.344	16.650
9325	R	*I. vomitoria*	0	0.1581	14.16	55.6	20.3	24.1	0.071	1.044	14.667
9325	T	*I. vomitoria*	1	0.182	19.13	68.3	13.6	18.1	0.120	2.091	17.371
9350	T	*I. vomitoria*	1	0.0905	18.1	57.6	20	22.4	0.077	0.837	10.801
9350	R	*I. vomitoria*	0	0.0655	24.41	62.6	19.3	18.1	0.171	2.599	15.168
9375	R	*I. vomitoria*	0	0.0825	16.24	58.4	28.3	13.3	0.075	1.145	15.196
9375	T	*I. vomitoria*	1	0.0866	12.52	67.6	15.7	16.7	0.135	2.259	16.687
9400	R	*I. vomitoria*	0	0.0952	10.56	62.1	6.6	31.3	0.123	2.198	17.842
9400	T	*I. vomitoria*	1	0.0906	11.03	76.7	12.6	10.7	0.240	4.813	20.069
9425	R	*I. vomitoria*	0	0.1595	9.14	71.3	16	12.7	0.073	1.042	14.313
9425	T	*I. vomitoria*	1	0.107	11.93	82.1	2.9	15	0.135	2.408	17.784
9450	R	*I. vomitoria*	0	0.0913	28.23	78.1	8.9	13	0.135	2.179	16.087
9475	R	*I. vomitoria*	0	0.074	31.31	65.3	14.3	20.4	0.200	2.090	10.438
9500	R	*I. vomitoria*	0	0.135	12.25	79.6	12	8.4	0.066	1.559	23.722
9525	R	*I. vomitoria*	0	0.1056	16.13	68.5	10.3	21.2	0.257	4.048	15.755
9550	T	*I. vomitoria*	1	0.091	12.66	38.4	39	22.6	0.327	5.498	16.791
9550	R	*I. vomitoria*	0	0.086	16.58	60.9	19.3	19.8	0.260	4.991	19.161
9575	R	*I. vomitoria*	0	0.1358	24.26	47	31.6	21.4	0.326	5.628	17.243
9600	R	*I. vomitoria*	0	0.084	27.25	69.2	13.2	17.6	0.208	3.947	18.985
9625	R	*I. vomitoria*	0	0.0983	15.26	63.6	16.6	19.8	0.178	3.883	21.844
9650	T	*I. vomitoria*	1	0.146	15.99	37	43.8	19.2	0.334	6.932	20.727
9650	R	*I. vomitoria*	0	0.0962	12.04	57.3	16.3	26.4	0.201	4.176	20.803
9675	R	*I. vomitoria*	0	0.125	8.94	65.5	13.7	20.8	0.378	6.589	17.428
9700	R	*I. vomitoria*	0	0.12	18.15	11	58.4	30.6	0.366	6.633	18.112
9725	R	*I. vomitoria*	0	0.0845	10.87	61.3	16.6	22.1	0.222	3.743	16.830
9750	R	*I. vomitoria*	0	0.0829	11.87	43.6	22.3	34.1	0.259	4.329	16.702
9775	R	*I. vomitoria*	0	0.082	22.14	48.2	20.3	31.5	0.192	3.044	15.848
9775	T	*I. vomitoria*	1	0.0703	14.06	53.9	20.3	25.8	0.174	3.151	18.097
9800	R	*I. vomitoria*	0	0.1081	29.32	60.7	12	27.3	0.228	3.361	14.770

Appendix A. Raw data from observational study, continued

9800	T	*I. vomitoria*	1	0.1072	15.5	43	22	35	0.266	5.543	20.864
9825	R	*I. vomitoria*	0	0.1041	30.47	48.1	20	31.9	0.189	3.015	15.958
9825	T	*I. vomitoria*	1	0.2961	35.1	15	59.5	25.5	0.322	6.107	18.940
9850	R	*I. vomitoria*	0	0.0845	29.94	66.1	12.8	21.1	0.206	3.814	18.526
9850	T	*I. vomitoria*	1	0.0785	17	57.3	11.3	31.4	0.206	3.814	18.526
9875	T	*I. vomitoria*	1	0.0898	27.08	14.2	36.6	49.2	0.193	2.852	14.760
9875	R	*I. vomitoria*	0	0.1145	21.32	29	26.3	44.7	0.319	5.466	17.130
9900	R	*I. vomitoria*	0	0.0981	26.04	12.5	32.3	55.2	0.232	4.045	17.429
9900	T	*I. vomitoria*	1	0.1	29.67	18.2	34.6	47.2	0.232	4.045	17.429
9925	R	*I. vomitoria*	0	0.19	18.57	49.6	16.3	34.1	0.392	6.610	16.879
9925	T	*I. vomitoria*	1	0.0964	18.33	7.2	40.6	52.2	0.148	3.042	20.556
9950	T	*I. vomitoria*	1	0.109	17.51	32.7	27.9	39.4	0.371	6.054	16.325
9950	R	*I. vomitoria*	0	0.0988	15.88	38.5	25.3	36.2	0.214	4.588	21.438
9975	R	*I. vomitoria*	0	0.0953	17.93	44.5	32.5	23	0.225	3.610	16.021
9975	T	*I. vomitoria*	1	0.1	17.84	43	22.3	34.7	0.212	3.933	18.592
10000	R	*I. vomitoria*	0	0.06	52.3	36.5	28.3	35.2	0.100	1.828	18.267
10000	T	*I. vomitoria*	1	0.0551	23.73	36.8	30.6	32.6	0.166	3.324	20.000
10025	R	*I. vomitoria*	0	0.0305	28.24	46.7	20.3	33	0.211	3.027	14.382
10025	T	*I. vomitoria*	1	0.1101	12.97	19.2	36.6	44.2	0.467	11.273	24.135
10075	R	*I. vomitoria*	0	0.04	33.01	53.1	20.6	26.3	0.247	5.176	20.971
10075	T	*I. vomitoria*	1	0.0376	35.6	35.6	34.3	30.1	0.196	4.536	23.136
10100	R	*I. vomitoria*	0	0.0528	27.8	17.2	50.4	32.4	0.289	3.897	13.495
10100	T	*I. vomitoria*	1	0.0472	32.34	54.5	17.3	28.2	0.295	6.241	21.175
10125	R	*I. vomitoria*	0	0.0207	21.43	52.38	18.42	29.2	0.089	1.343	15.163
10125	T	*I. vomitoria*	1	0.04	22.06	64.1	12.3	23.6	0.231	3.751	16.269
10150	T	*I. vomitoria*	1	0.0442	22.4	61.9	18.3	19.8	0.134	2.115	15.799
10150	R	*I. vomitoria*	0	0.0553	21.61	29.8	46.4	23.8	0.417	7.319	17.544
10175	R	*I. vomitoria*	0	0.0645	21.96	65.6	18	16.4	0.243	3.827	15.767
10175	T	*I. vomitoria*	1	0.059	21.97	53.9	21.7	24.4	0.213	3.486	16.371
10200	R	*I. vomitoria*	0	0.0403	28.84	43	26.3	30.7	0.227	3.117	13.733
10200	T	*I. vomitoria*	1	0.0465	25.25	53.9	27	19.1	0.216	3.614	16.707
10225	R	*I. vomitoria*	0	0.0522	32.16	48.9	25.3	25.8	0.170	2.776	16.304
10225	T	*I. vomitoria*	1	0.0456	20.41	61	12.3	26.7	0.121	2.256	18.719
10250	T	*I. vomitoria*	1	0.0256	29.96	43.1	32.3	24.6	0.143	2.123	14.855
10250	R	*I. vomitoria*	0	0.036	30.71	47.6	26.3	26.1	0.213	3.222	15.105
10275	T	*I. vomitoria*	1	0.041	31.38	33.9	35.3	30.8	0.167	2.449	14.652

Appendix A. Raw data from observational study, continued

10275	R	*I. vomitoria*	0	0.0313	30.26	32.5	30.3	37.2	0.151	2.570	16.968
10300	R	*I. vomitoria*	0	0.0433	39.19	52.4	20.6	27	0.211	3.061	14.472
10300	T	*I. vomitoria*	1	0.0363	37.4	34.8	30.6	34.6	0.154	2.425	15.737
10325	R	*I. vomitoria*	0	0.045	34.91	34.7	30.6	34.7	0.223	2.589	11.625
10325	T	*I. vomitoria*	1	0.0335	29.68	44.1	26	29.9	0.210	2.496	11.890
10350	R	*I. vomitoria*	0	0.0287	52.01	42.5	26.3	31.2	0.117	1.486	12.669
10350	T	*I. vomitoria*	1	0.0344	45.49	47.1	24.6	28.3	0.165	2.664	16.148
10375	T	*I. vomitoria*	1	0.0308	48.15	44.8	26.9	28.3	0.115	1.516	13.167
10375	R	*I. vomitoria*	0	0.066	50.84	53.3	25.7	21	0.189	2.834	14.966
10400	T	*I. vomitoria*	1	0.0475	56.49	50.8	28.9	20.3	0.110	1.406	12.829
10400	R	*I. vomitoria*	0	0.0785	48.61	36.2	34.3	29.5	0.341	5.620	16.470
10425	R	*I. vomitoria*	0	0.073	67.26	43.4	32	24.6	0.139	1.909	13.765
10425	T	*I. vomitoria*	1	0.0778	61.64	37.1	30.3	32.6	0.185	2.598	14.022
10450	R	*I. vomitoria*	0	0.0466	73.87	50.8	22.72	26.48	0.086	0.962	11.244
10450	T	*I. vomitoria*	1	0.0533	65.18	49.9	26.3	23.8	0.163	2.505	15.379
10475	T	*I. vomitoria*	1	0.0459	70.19	43.3	34	22.7	0.102	1.218	11.894
10475	R	*I. vomitoria*	0	0.0455	57.57	42.8	30.6	26.6	0.099	1.228	12.465
10500	T	*I. vomitoria*	1	0.0591	61.95	49.1	29.3	21.6	0.107	1.371	12.825
10500	R	*I. vomitoria*	0	0.0411	54.45	45.3	30.3	24.4	0.093	1.269	13.571
10525	T	*I. vomitoria*	1	0.0497	64.95	64.1	18	17.9	0.136	1.855	13.609
10525	R	*I. vomitoria*	0	0.0485	72.95	62.2	21.4	16.4	0.099	1.362	13.733
10550	T	*I. vomitoria*	1	0.045	75.19	75.5	12.9	11.6	0.134	1.725	12.866
10550	R	*I. vomitoria*	0	0.044	61.7	57.6	23.7	18.7	0.097	1.259	12.939
10575	R	*I. vomitoria*	0	0.0346	78.04	56.1	24	19.9	0.110	1.476	13.356
10575	T	*I. vomitoria*	1	0.0454	62.31	52.8	22.6	24.6	0.099	1.450	14.682
10600	R	*I. vomitoria*	0	0.0446	82.43	36.5	34.3	29.2	0.126	1.540	12.237
10600	T	*I. vomitoria*	1	0.0516	78.77	75.6	9.7	14.7	0.141	1.951	13.854
10625	T	*I. vomitoria*	1	0.082	80.74	49.1	25.3	25.6	0.189	2.385	12.605
10625	R	*I. vomitoria*	0	0.0493	71.82	71.5	16	12.5	0.130	1.871	14.358
10650	R	*I. vomitoria*	0	0.024	82.84	70.7	6	23.3	0.146	1.476	10.121
10650	T	*I. vomitoria*	1	0.0261	66.11	69.3	16.3	14.4	0.127	1.830	14.378
10675	T	*I. vomitoria*	1	0.0257	77.59	63.6	14	22.4	0.174	2.314	13.303
10675	R	*I. vomitoria*	0	0.0258	66.14	74.7	4	21.3	0.102	1.385	13.600
10700	T	*I. vomitoria*	1	0.016	73.46	59.5	16.6	23.9	0.086	1.032	11.966
10700	R	*I. vomitoria*	0	0.031	76.02	58.2	16.6	25.2	0.092	1.142	12.463
10725	T	*I. vomitoria*	1	0.03	75.56	33.8	27.3	38.9	0.158	2.269	14.372

Appendix A. Raw data from observational study, continued

10725	R	*I. vomitoria*	0	0.0228	66.72	38.8	37.3	23.9	0.150	2.161	14.384
10750	R	*I. vomitoria*	0	0.0344	62.6	75.5	7.7	16.8	0.129	1.839	14.299
10750	T	*I. vomitoria*	1	0.0418	67.38	37.3	18	44.7	0.159	2.327	14.645
10775	R	*I. vomitoria*	0	0.023	59.38	30.8	30.6	38.6	0.160	2.164	13.525
10775	T	*I. vomitoria*	1	0.2441	65.33	24.5	53.9	21.6	0.200	2.899	14.495
10800	T	*I. vomitoria*	1	0.0275	76.33	26.2	36.6	37.2	0.189	2.095	11.081
10800	R	*I. vomitoria*	0	0.0436	75.6	34.7	30.3	35	0.219	3.114	14.206
10825	T	*I. vomitoria*	1	0.0407	78.22	34.5	33.3	32.2	0.164	2.095	12.756
10825	R	*I. vomitoria*	0	0.0291	82.53	43.2	26.3	30.5	0.185	2.380	12.843
10850	T	*I. vomitoria*	1	0.0477	79.22	32.5	38.3	29.2	0.171	1.993	11.635
10850	R	*I. vomitoria*	0	0.039	78.25	26.1	37.3	36.6	0.141	1.808	12.855
10875	R	*I. vomitoria*	0	0.0392	77.51	36.8	30.6	32.6	0.218	2.423	11.116
10875	T	*I. vomitoria*	1	0.0525	82.25	36.7	36.9	26.4	0.206	2.519	12.199
10900	R	*I. vomitoria*	0	0.04	83.19	35	34.6	30.4	0.285	3.128	10.972
10900	T	*I. vomitoria*	1	0.0382	86.68	40.7	30.3	29	0.145	1.683	11.635
10925	R	*I. vomitoria*	0	0.0394	87.64	62.5	18	19.5	0.113	1.812	16.084
10950	R	*I. vomitoria*	0	0.037	88.3	76.1	10	13.9	0.104	1.329	12.814
10975	R	*I. vomitoria*	0	0.0307	87.67	61.3	18.3	20.4	0.124	1.369	11.056
11000	T	*I. vomitoria*	1	0.02	88.3	71	10.6	18.4	0.122	1.298	10.636
11000	R	*I. vomitoria*	0	0.029	87.68	50.1	24.3	25.6	0.155	2.323	15.020
11025	R	*I. vomitoria*	0	0.0232	86.4	83.3	10.3	6.4	0.040	0.511	12.761
11025	T	*I. vomitoria*	1	0.037	86.66	58.5	18.3	23.2	0.210	2.919	13.924
11050	T	*I. vomitoria*	1	0.0328	82.2	80.1	14.1	5.8	0.215	3.137	14.586
11050	R	*I. vomitoria*	0	0.0262	85.57	82.2	10	7.8	0.115	1.760	15.250
11075	R	*I. vomitoria*	0	0.019	85.62	73.3	4.6	22.1	0.061	0.503	8.178
11100	R	*I. vomitoria*	0	0.0227	87.24	83.2	11.2	5.6	0.069	0.581	8.399
11100	T	*I. vomitoria*	1	0.0234	85.75	81.5	12.6	5.9	0.091	0.944	10.423
11125	T	*I. vomitoria*	1	0.0221	85.89	36.5	36.3	27.2	0.150	1.910	12.726
11125	R	*I. vomitoria*	0	0.0306	85.4	44.5	28.9	26.6	0.147	2.111	14.316
11150	T	*I. vomitoria*	1	0.0242	80.25	68.5	22.3	9.2	0.167	1.773	10.638
11150	R	*I. vomitoria*	0	0.03	84.44	51.3	29.3	19.4	0.189	3.501	18.529
11175	R	*I. vomitoria*	0	0.0256	85.92	61.5	18.9	19.6	0.144	1.762	12.241
11175	T	*I. vomitoria*	1	0.0369	80.18	52.2	20.6	27.2	0.162	2.004	12.359
11200	T	*I. vomitoria*	1	0.028	85.85	34.8	34.6	30.6	0.214	2.535	11.837
11200	R	*I. vomitoria*	0	0.0295	85.75	48.4	28.9	22.7	0.182	2.480	13.643
11225	R	*I. vomitoria*	0	0.06	81.15	55.3	26.3	18.4	0.158	1.960	12.395

Appendix A. Raw data from observational study, continued

11250	T	*I. vomitoria*	1	0.0555	84.58	32.2	37.2	30.6	0.112	1.544	13.776	
11250	R	*I. vomitoria*	0	0.0716	75.79	23.1	40.6	36.3	0.197	2.841	14.446	
11275	T	*I. vomitoria*	1	0.07	63.58	45.5	31.2	23.3	0.261	4.069	15.571	
11275	R	*I. vomitoria*	0	0.1037	74.05	65.8	14.9	19.3	0.147	2.854	19.361	
11300	R	*I. vomitoria*	0	0.0814	57.24	33.5	34.3	32.2	0.320	4.026	12.584	
11300	T	*I. vomitoria*	1	0.0804	64.25	51.6	25.7	22.7	0.288	4.385	15.228	
0	R	*M. cerifera*	0	6.66	98.15	73.6	10.6	15.8	0.201	3.136	15.616	
100	R	*M. cerifera*	0	7.53	98.15	30.4	43	26.6	0.386	4.648	12.035	
200	R	*M. cerifera*	0	9.4	98.15	61.8	18.3	19.9	0.373	5.527	14.819	
300	R	*M. cerifera*	0	9.03	98.15	59.1	30.9	10	0.459	5.935	12.939	
400	R	*M. cerifera*	0	7.4	98.15	13.8	43.6	42.6	0.291	4.170	14.353	
500	R	*M. cerifera*	0	9.28	98.15	56.8	16.9	26.3	0.173	2.682	15.472	
600	R	*M. cerifera*	0	8.49	98.15	57.1	18.3	24.6	0.324	4.831	14.893	
700	R	*M. cerifera*	0	3.567	98.15	67	14	19	0.047	0.424	8.967	
800	R	*M. cerifera*	0	8.59	98.15	34.4	35	30.6	0.253	4.045	15.992	
900	R	*M. cerifera*	0	10.88	98.15	35	38.4	26.6	0.318	4.948	15.583	
1000	R	*M. cerifera*	0	7.38	98.15	69.5	18.6	11.9	0.280	4.108	14.654	
1100	R	*M. cerifera*	0	10.1	98.15	27	39	34	0.289	3.897	13.495	
1200	R	*M. cerifera*	0	11.53	98.15	.	.	.	0.451	6.415	14.211	
1300	R	*M. cerifera*	0	8.94	98.15	74.4	6.4	19.2	0.467	6.793	14.551	
1400	R	*M. cerifera*	0	6.43	98.15	59.5	24.3	16.2	0.215	2.749	12.772	
1500	R	*M. cerifera*	0	10	98.15	37	52	11	0.436	5.976	13.693	
1600	R	*M. cerifera*	0	8.34	98.15	73.2	12.3	14.5	0.411	6.091	14.814	
1700	R	*M. cerifera*	0	7.3	98.15	30.4	51	18.6	0.272	3.751	13.791	
1800	R	*M. cerifera*	0	9	98.15	29	49.7	21.3	0.317	4.648	14.678	
1900	R	*M. cerifera*	0	8	98.15	43	37	20	0.282	4.293	15.245	
2000	R	*M. cerifera*	0	11.7	98.15	32.4	27.2	40.4	0.381	5.262	13.821	
2100	R	*M. cerifera*	0	8.75	98.15	67.6	16.3	16.1	0.154	2.002	12.975	
2200	R	*M. cerifera*	0	11.63	98.15	13.8	26.4	59.8	0.284	3.950	13.903	
2300	R	*M. cerifera*	0	5.7	98.15	39.4	36.3	24.3	0.095	1.060	11.213	
2400	R	*M. cerifera*	0	8.725	98.15	67.1	10.3	22.6	0.375	6.057	16.170	
2500	R	*M. cerifera*	0	10.28	98.15	65.1	15.4	19.5	0.312	5.125	16.407	
2600	R	*M. cerifera*	0	9.76	98.15	59.5	21.2	19.3	0.324	4.831	14.893	
2700	R	*M. cerifera*	0	9.66	98.15	70.9	13.2	15.9	0.386	5.829	15.113	
2800	R	*M. cerifera*	0	10.5	98.15	67.2	11.5	21.3	0.345	5.384	15.610	
2900	R	*M. cerifera*	0	5.76	98.15	59.5	18.9	21.6	0.217	2.588	11.923	

Appendix A. Raw data from observational study, continued

3000	R	*M. cerifera*	0	9.19	98.15	39.6	39	21.4	0.389	5.392	13.861
3100	R	*M. cerifera*	0	7.91	98.15	64.1	22	13.9	0.364	5.916	16.244
3200	R	*M. cerifera*	0	8.5	98.15	57.5	20.6	21.9	0.263	3.985	15.147
3300	R	*M. cerifera*	0	9.3	98.15	45.8	19.6	34.6	0.281	4.065	14.476
3400	R	*M. cerifera*	0	7.11	98.15	60.5	31.2	8.3	0.433	6.931	15.991
3500	R	*M. cerifera*	0	7.1	98.15	64.1	18.6	17.3	0.446	6.784	15.200
3600	R	*M. cerifera*	0	9.668	98.15	49.1	40.6	10.3	0.472	6.799	14.396
3700	R	*M. cerifera*	0	8.72	98.15	39	33.2	27.8	0.208	2.956	14.241
3800	R	*M. cerifera*	0	13	98.15	67.5	11.2	21.3	0.418	7.770	18.566
3900	R	*M. cerifera*	0	15.43	98.15	48.9	39.8	11.3	0.464	9.034	19.483
4000	R	*M. cerifera*	0	14.6	98.15	59.5	25.5	15	1.079	11.161	10.342
4100	R	*M. cerifera*	0	18.46	98.15	56.8	29.9	13.3	0.482	8.177	16.957
4200	R	*M. cerifera*	0	16.25	97.87	68.8	19.3	11.9	0.420	9.002	21.415
4300	R	*M. cerifera*	0	6.07	97.69	59.9	18.9	21.2	0.151	3.122	20.669
4400	R	*M. cerifera*	0	3.93	90.81	51.6	28	20.4	0.079	1.297	16.380
4500	R	*M. cerifera*	0	3.9	89.67	39.2	42.7	18.1	0.060	0.240	4.003
4600	R	*M. cerifera*	0	4.72	88.53	37	35.2	27.8	0.148	1.914	12.927
4625	R	*M. cerifera*	0	3.891	85.58	52.2	22.3	25.5	0.071	1.055	14.817
4650	R	*M. cerifera*	0	0.892	73.73	57.3	26.3	16.4	0.068	0.899	13.266
4675	T	*M. cerifera*	1	0.84	59.55	79.3	10.3	10.4	0.064	0.743	11.551
4675	R	*M. cerifera*	0	0.4494	60.46	81.5	7.2	11.3	0.068	1.037	15.159
4700	T	*M. cerifera*	1	0.5	50.59	67.6	18.3	14.1	0.081	0.826	10.176
4700	R	*M. cerifera*	0	0.463	59.45	63.3	14.3	22.4	0.091	1.229	13.512
4725	R	*M. cerifera*	0	0.2	58.7	57.9	24.6	17.5	0.106	0.957	9.032
4725	T	*M. cerifera*	1	0.556	53.74	73.9	14	12.1	0.127	2.307	18.169
4750	R	*M. cerifera*	0	0.3425	43.58	35.8	36	28.2	0.083	0.702	8.454
4775	T	*M. cerifera*	1	0.487	41.33	38.6	43.3	18.1	0.097	1.676	17.210
4775	R	*M. cerifera*	0	0.2601	40.31	59	20.3	20.7	0.149	2.775	18.569
4800	R	*M. cerifera*	0	0.3266	40.16	76.4	11.8	11.8	0.053	0.694	13.180
4800	T	*M. cerifera*	1	0.2054	36.35	78.7	6.6	14.7	0.063	0.851	13.500
4825	T	*M. cerifera*	1	0.3992	43.7	75.9	5.4	18.7	0.056	0.738	13.169
4825	R	*M. cerifera*	0	2.652	48.08	49	22	29	0.060	0.967	16.123
4850	R	*M. cerifera*	0	1.354	71.21	51	21.7	27.3	0.127	1.887	14.864
4850	T	*M. cerifera*	1	0.435	52.18	58.2	16.6	25.2	0.083	1.337	16.090
4875	R	*M. cerifera*	0	1.5774	79.52	54.4	28.5	17.1	0.114	1.741	15.319
4900	R	*M. cerifera*	0	3.735	85.12	10.2	68	21.8	0.181	3.492	19.271

Appendix A. Raw data from observational study, continued

4925	R	*M. cerifera*	0	4.35	84.05	64.7	17.8	17.5	0.379	6.877	18.142	
4950	R	*M. cerifera*	0	4.75	81.5	77.3	13.5	9.2	0.556	8.788	15.808	
4975	R	*M. cerifera*	0	2.28	83.92	52.7	19.7	27.6	0.159	3.044	19.125	
5000	R	*M. cerifera*	0	1.012	84.23	79.5	3.2	17.3	0.153	2.412	15.736	
5025	R	*M. cerifera*	0	2.025	85.44	59	20.9	20.1	0.103	1.951	18.928	
5050	R	*M. cerifera*	0	1.4	79.42	56.1	20.6	23.3	0.174	2.207	12.677	
5075	R	*M. cerifera*	0	1.444	71.6	53	20.9	26.1	0.104	1.477	14.220	
5100	R	*M. cerifera*	0	0.4307	48.14	61.6	14.3	24.1	0.105	1.699	16.182	
5100	T	*M. cerifera*	1	0.655	57.21	18.4	47.6	34	0.131	2.584	19.669	
5125	T	*M. cerifera*	1	0.448	41.51	67.3	16	16.7	0.069	0.988	14.320	
5125	R	*M. cerifera*	0	0.332	48.52	59	14	27	0.123	1.952	15.829	
5150	T	*M. cerifera*	1	0.603	41.32	69.3	18	12.7	0.079	0.998	12.621	
5150	R	*M. cerifera*	0	0.547	46.11	59.6	18.3	22.1	0.124	1.595	12.843	
5175	R	*M. cerifera*	0	0.4485	42.44	68.4	16.6	15	0.076	1.050	13.791	
5175	T	*M. cerifera*	1	0.368	42.98	76.1	12	11.9	0.050	0.954	18.947	
5200	T	*M. cerifera*	1	0.584	38.27	72.1	12	15.9	0.097	0.989	10.163	
5200	R	*M. cerifera*	0	0.4485	40.58	65	16.3	18.7	0.062	1.059	17.190	
5225	R	*M. cerifera*	0	1.601	70.58	45	18	37	0.244	3.834	15.703	
5250	R	*M. cerifera*	0	1.788	70.47	59.6	18.3	22.1	0.243	2.772	11.408	
5275	R	*M. cerifera*	1	0.2601	39.15	65	12.3	22.7	0.130	2.107	16.204	
5275	T	*M. cerifera*	1	0.2943	47.74	61	12.3	26.7	0.271	4.810	17.739	
5300	T	*M. cerifera*	1	0.162	40.02	61.3	12.6	26.1	0.075	1.230	16.344	
5300	R	*M. cerifera*	0	0.3201	43.3	57.6	18.3	24.1	0.103	1.755	17.083	
5325	T	*M. cerifera*	1	0.1754	44.81	63.4	18.2	18.4	0.087	1.288	14.812	
5325	R	*M. cerifera*	0	0.18	35.6	66.1	12	21.9	0.078	1.433	18.342	
5350	T	*M. cerifera*	1	0.1209	47.88	67	16.3	16.7	0.075	1.086	14.517	
5350	R	*M. cerifera*	0	0.1334	33.62	62.7	16.6	20.7	0.092	1.584	17.249	
5375	R	*M. cerifera*	0	0.458	43.7	63.9	14.6	21.5	0.078	1.493	19.139	
5375	T	*M. cerifera*	1	0.1735	38.11	54.7	24.3	21	0.068	1.360	20.134	
5400	R	*M. cerifera*	0	0.176	28.58	67.5	17	15.5	0.106	1.381	12.969	
5425	R	*M. cerifera*	0	0.1485	58.85	56.7	16.9	26.4	0.095	1.270	13.383	
5450	R	*M. cerifera*	0	0.1672	63.7	65.4	20.3	14.3	0.129	1.665	12.887	
5475	R	*M. cerifera*	0	0.166	70.8	6	35.4	58.6	0.092	1.197	13.039	
5500	R	*M. cerifera*	0	0.3688	60.92	52.4	16	31.6	0.133	1.420	10.686	
5500	T	*M. cerifera*	1	0.244	61.82	67.3	9.4	23.3	0.086	1.235	14.420	
5525	T	*M. cerifera*	1	0.2152	55.2	60.2	16.3	23.5	0.066	0.708	10.742	

Appendix A. Raw data from observational study, continued

5525	R	*M. cerifera*	0	0.3747	38.75	61.3	10.6	28.1	0.106	1.742	16.488	
5550	R	*M. cerifera*	0	0.1422	46.2	57.9	16.6	25.5	0.101	1.483	14.649	
5575	T	*M. cerifera*	1	0.178	49.26	67	14	19	0.123	1.759	14.269	
5575	R	*M. cerifera*	0	0.15	54.43	65.5	14.6	19.9	0.115	1.678	14.581	
5600	T	*M. cerifera*	1	0.1346	39.87	70.4	10.3	19.3	0.125	1.698	13.621	
5600	R	*M. cerifera*	0	0.1679	58.72	75.5	8	16.5	0.130	2.139	16.404	
5625	R	*M. cerifera*	0	0.2068	56.13	63.9	10.6	25.5	0.143	1.713	11.971	
5650	R	*M. cerifera*	0	0.2255	66.55	65.9	14.3	19.8	0.102	1.703	16.733	
5675	R	*M. cerifera*	0	0.129	66.96	65.3	12.3	22.4	0.101	1.391	13.779	
5675	T	*M. cerifera*	1	0.1156	61.09	72	8	20	0.107	1.685	15.708	
5700	R	*M. cerifera*	0	0.2276	65.18	60.7	16.3	23	0.106	1.293	12.170	
5725	R	*M. cerifera*	0	0.21	56.41	60.1	14	25.9	0.100	1.646	16.533	
5750	T	*M. cerifera*	1	0.23	51.4	51.08	23.42	25.5	0.154	1.608	10.459	
5750	R	*M. cerifera*	0	0.285	63.33	61.5	15.2	23.3	0.093	1.456	15.726	
5775	R	*M. cerifera*	0	0.2423	44.92	64.7	12.3	23	0.106	1.565	14.828	
5775	T	*M. cerifera*	1	0.376	35.85	57.1	20.3	22.6	0.093	1.810	19.458	
5800	R	*M. cerifera*	0	0.4027	24.3	63.3	16.6	20.1	0.091	1.594	17.593	
5800	T	*M. cerifera*	1	0.79	22.76	45.3	32.3	22.4	0.098	1.952	19.934	
5825	R	*M. cerifera*	0	0.51	33.81	64.4	14.3	21.3	0.100	1.709	17.137	
5825	T	*M. cerifera*	1	0.397	38.28	55.5	21.2	23.3	0.188	3.436	18.260	
5850	T	*M. cerifera*	1	0.3274	49.55	64.1	14.6	21.3	0.261	4.255	16.279	
5850	R	*M. cerifera*	0	1.022	52.6	6.4	55	38.6	0.247	4.514	18.267	
5875	T	*M. cerifera*	1	0.353	41.29	66.1	10.3	23.6	0.167	3.350	20.020	
5875	R	*M. cerifera*	0	0.4	43.63	65.9	17	17.1	0.155	3.460	22.365	
5900	T	*M. cerifera*	1	0.2699	39.33	59	17.7	23.3	0.151	3.382	22.373	
5900	R	*M. cerifera*	0	0.4885	37.75	52.7	22.6	24.7	0.085	1.918	22.645	
5925	R	*M. cerifera*	0	0.2638	43.77	54.7	22.3	23	0.302	5.779	19.113	
5925	T	*M. cerifera*	1	0.2638	43.77	56.7	20.6	22.7	0.167	3.588	21.523	
5950	R	*M. cerifera*	0	0.383	36.58	65.9	19.7	14.4	0.211	3.764	17.875	
5975	T	*M. cerifera*	1	0.509	36.77	60.4	16.6	23	0.116	1.692	14.612	
5975	R	*M. cerifera*	0	0.95	58.57	42.7	22.6	34.7	0.237	4.621	19.496	
6000	T	*M. cerifera*	1	0.5	42.84	62.1	16.6	21.3	0.137	2.022	14.744	
6000	R	*M. cerifera*	0	0.2055	43.54	63.3	23.5	13.2	0.164	3.119	18.976	
6025	T	*M. cerifera*	1	0.2427	47.48	51.3	22.3	26.4	0.094	1.217	12.915	
6025	R	*M. cerifera*	0	0.166	44.25	51	22.3	26.7	0.196	3.024	15.453	
6050	T	*M. cerifera*	1	0.196	45.54	45.9	23.6	30.5	0.145	2.110	14.506	

Appendix A. Raw data from observational study, continued

6050	R	*M. cerifera*	0	0.2883	54.92	50.7	20.9	28.4	0.186	2.846	15.286	
6075	R	*M. cerifera*	0	0.2682	58.48	63.3	16.9	19.8	0.171	2.937	17.131	
6100	R	*M. cerifera*	0	0.25	57.08	28.6	35.3	36.1	0.141	2.109	14.969	
6100	T	*M. cerifera*	1	0.1211	56.89	45.3	25.7	29	0.083	1.517	18.222	
6125	T	*M. cerifera*	1	0.1581	56.56	47	26.3	26.7	0.157	2.420	15.439	
6125	R	*M. cerifera*	0	0.3457	52.6	40.2	28.3	31.5	0.148	2.560	17.273	
6150	R	*M. cerifera*	0	0.48	38.27	51.9	24.6	23.5	0.064	0.871	13.596	
6150	T	*M. cerifera*	1	0.2967	35.64	57.5	21.2	21.3	0.096	1.670	17.421	
6175	T	*M. cerifera*	1	0.4254	33.62	73	6.9	20.1	0.090	1.357	15.059	
6175	R	*M. cerifera*	0	0.287	23.89	56.1	16.9	27	0.132	2.041	15.442	
6200	R	*M. cerifera*	0	0.1182	47.55	37.9	32.6	29.5	0.091	1.160	12.728	
6200	T	*M. cerifera*	1	0.2036	43.23	57.2	15.2	27.6	0.108	1.654	15.257	
6225	R	*M. cerifera*	0	0.238	37.33	42.4	35.5	22.1	0.072	0.962	13.401	
6225	T	*M. cerifera*	1	0.3472	37.32	54.1	22	23.9	0.182	2.890	15.883	
6250	R	*M. cerifera*	0	0.2337	35.92	50.7	26.6	22.7	0.088	1.361	15.546	
6250	T	*M. cerifera*	1	0.372	35.15	51	28.3	20.7	0.091	1.643	18.113	
6275	R	*M. cerifera*	0	0.2557	34.76	63.8	22.3	13.9	0.121	1.803	14.960	
6275	T	*M. cerifera*	1	0.2112	32.93	60.6	21	18.4	0.060	0.902	14.960	
6300	R	*M. cerifera*	0	0.158	32.99	61.3	20	18.7	0.095	1.068	11.232	
6300	T	*M. cerifera*	1	0.2402	35.84	63	20	17	0.074	1.082	14.620	
6325	T	*M. cerifera*	1	0.2056	27.7	64.9	7.5	27.6	0.130	1.786	13.726	
6325	R	*M. cerifera*	0	0.091	42.5	64.7	12.6	22.7	0.147	2.045	13.955	
6350	R	*M. cerifera*	0	0.1576	49.81	55.8	14.9	29.3	0.178	2.376	13.379	
6375	R	*M. cerifera*	0	0.2454	49.58	31.3	36.3	32.4	0.159	2.341	14.765	
6400	R	*M. cerifera*	0	0.1919	51.6	15.3	48	36.7	0.185	2.569	13.887	
6425	R	*M. cerifera*	0	0.1511	50.67	39.5	26.6	33.9	0.147	2.009	13.661	
6450	R	*M. cerifera*	0	0.3422	44.39	41	28.3	30.7	0.162	2.440	15.055	
6450	T	*M. cerifera*	1	0.199	44.77	55.3	18.3	26.4	0.104	1.869	18.027	
6475	R	*M. cerifera*	0	0.2442	34.34	50.1	26	23.9	0.152	2.267	14.933	
6500	R	*M. cerifera*	0	0.0902	33.79	50.7	18	31.3	0.128	2.176	16.996	
6500	T	*M. cerifera*	1	0.135	33.26	65.3	16.6	18.1	0.120	2.449	20.351	
6525	T	*M. cerifera*	1	0.211	39.42	77.3	16.3	6.4	0.120	1.804	15.070	
6525	R	*M. cerifera*	0	0.1511	22.51	53.5	19.2	27.3	0.122	2.243	18.387	
6550	R	*M. cerifera*	0	0.1639	24.73	53	20.3	26.7	0.117	2.069	17.749	
6550	T	*M. cerifera*	1	0.541	31.72	65.6	20	14.4	0.087	1.770	20.309	
6575	R	*M. cerifera*	0	0.2563	32.71	63.6	18.3	18.1	0.080	1.066	13.393	

Appendix A. Raw data from observational study, continued

6575	T	*M. cerifera*	1	0.4351	37.03	61.3	22	16.7	0.058	0.965	16.612
6600	R	*M. cerifera*	0	0.165	27.81	75.3	13.2	11.5	0.084	0.989	11.721
6600	T	*M. cerifera*	1	0.1482	32.89	55	26.3	18.7	0.060	1.198	19.825
6625	T	*M. cerifera*	1	0.1645	29.79	72.1	10.3	17.6	0.172	1.970	11.480
6625	R	*M. cerifera*	0	0.3356	19.7	67.9	22	10.1	0.103	1.512	14.713
6650	T	*M. cerifera*	1	0.2161	19.42	78.1	10	11.9	0.154	2.162	14.028
6650	R	*M. cerifera*	0	0.181	22.84	19	51	30	0.205	3.380	16.522
6675	R	*M. cerifera*	0	0.1014	32.82	43.6	20	36.4	0.093	1.486	15.936
6675	T	*M. cerifera*	1	0.2406	30.4	65.3	26	8.7	0.129	2.836	21.923
6700	R	*M. cerifera*	0	0.1347	35.39	62.4	14.9	22.7	0.104	0.940	9.067
6700	T	*M. cerifera*	1	0.3213	26.37	69.9	16	14.1	0.068	1.285	18.963
6725	R	*M. cerifera*	0	0.1184	35.37	77	4.3	18.7	0.127	2.274	17.859
6750	R	*M. cerifera*	0	0.1147	39.63	65.6	12.3	22.1	0.157	1.890	12.027
6775	R	*M. cerifera*	0	0.22	43.78	74.7	14.9	10.4	0.064	0.734	11.390
6775	T	*M. cerifera*	1	0.2376	39.65	63.4	16.9	19.7	0.054	0.812	15.132
6800	R	*M. cerifera*	0	0.2425	40.82	56.7	16.3	27	0.071	1.034	14.602
6800	T	*M. cerifera*	1	0.1973	44.76	55.6	22.6	21.8	0.133	2.136	16.072
6825	T	*M. cerifera*	1	0.179	46.5	47	26.6	26.4	0.102	1.442	14.143
6825	R	*M. cerifera*	0	0.1897	44.68	54.1	16.9	29	0.081	1.267	15.628
6850	T	*M. cerifera*	1	0.1231	38.71	45.6	22	32.4	0.119	1.488	12.538
6850	R	*M. cerifera*	0	0.1405	40.71	49.4	16.9	33.7	0.091	1.280	13.989
6875	R	*M. cerifera*	0	0.129	46.23	56.2	20	23.8	0.176	2.872	16.319
6875	T	*M. cerifera*	1	0.1318	43.03	51	18.3	30.7	0.167	2.951	17.625
6900	T	*M. cerifera*	1	0.2135	43.26	55.1	22.6	22.3	0.118	1.906	16.084
6900	R	*M. cerifera*	0	.	49.35	.	.	.	.	.	.
6925	T	*M. cerifera*	1	0.2568	38.84	73.6	11.7	14.7	0.204	2.543	12.465
6925	R	*M. cerifera*	0	0.2054	30.22	14.4	58.4	27.2	0.180	3.714	20.666
6950	R	*M. cerifera*	0	0.1992	40.49	70.1	7.2	22.7	0.181	2.198	12.143
6950	T	*M. cerifera*	1	0.234	34.78	69.1	13.9	17	0.104	1.829	17.537
6975	R	*M. cerifera*	0	0.133	55.2	57.5	9.2	33.3	0.195	2.527	12.939
7000	R	*M. cerifera*	0	0.3326	49.85	74.1	8.3	17.6	0.200	4.356	21.825
7025	T	*M. cerifera*	1	0.1204	43.1	76.1	6.3	17.6	0.175	3.378	19.263
7025	R	*M. cerifera*	0	0.1636	56.56	56.4	18.9	24.7	0.162	3.303	20.339
7050	R	*M. cerifera*	0	0.1195	38.45	71.9	11.7	16.4	0.113	2.075	18.369
7075	R	*M. cerifera*	0	0.1926	44.32	64.4	8.9	26.7	0.172	3.139	18.224
7100	T	*M. cerifera*	1	0.128	46.49	36.7	36.6	26.7	0.169	2.462	14.607

Appendix A. Raw data from observational study, continued

7100	R	*M. cerifera*	0	0.1135	33.72	47	18.3	34.7	0.165	3.255	19.690	
7125	R	*M. cerifera*	0	0.1994	47.04	53.3	16.6	30.1	0.184	2.879	15.639	
7150	R	*M. cerifera*	0	0.1675	36.65	55	18.9	26.1	0.096	1.699	17.763	
7175	R	*M. cerifera*	0	0.1323	36.82	42.2	24.6	33.2	0.153	2.044	13.343	
7175	T	*M. cerifera*	1	0.1663	39.18	37	28.3	34.7	0.207	3.213	15.535	
7200	R	*M. cerifera*	0	0.11	47.93	41	28.6	30.4	0.156	2.471	15.850	
7225	R	*M. cerifera*	0	0.088	55.98	39	29.6	31.4	0.158	2.289	14.524	
7225	T	*M. cerifera*	1	0.0833	55.97	15	43	42	0.124	2.131	17.165	
7250	R	*M. cerifera*	0	0.1118	50.11	37	32.3	30.7	0.139	2.051	14.719	
7250	T	*M. cerifera*	1	0.1581	47	34.7	29.2	36.1	0.154	2.756	17.920	
7275	R	*M. cerifera*	0	0.1181	37.5	36.1	30.6	33.3	0.169	2.232	13.223	
7275	T	*M. cerifera*	1	0.4203	23.71	44.7	28.3	27	0.129	1.975	15.314	
7300	T	*M. cerifera*	1	0.129	21.48	51	24.6	24.4	0.158	2.315	14.653	
7300	R	*M. cerifera*	0	0.115	21.99	59.7	19.9	20.4	0.133	2.063	15.558	
8200	T	*M. cerifera*	1	0.141	22.88	51.6	21.7	26.7	0.119	1.857	15.652	
8200	R	*M. cerifera*	0	0.1064	21.69	61	22.3	16.7	0.060	1.083	18.098	
8225	R	*M. cerifera*	0	0.1306	14.91	57	20.6	22.4	0.053	1.129	21.213	
8250	T	*M. cerifera*	1	0.089	35.98	76.7	6.6	16.7	0.050	0.689	13.850	
8250	R	*M. cerifera*	0	0.0823	14.04	70.1	8.3	21.6	0.093	1.500	16.089	
8275	R	*M. cerifera*	0	0.0496	36.55	74.1	12	13.9	0.060	0.826	13.771	
8275	T	*M. cerifera*	1	0.0492	44.77	67	23	10	0.054	1.173	21.558	
8300	R	*M. cerifera*	0	0.0504	52.86	74.4	11.5	14.1	0.098	1.482	15.092	
8325	R	*M. cerifera*	0	0.0482	58.2	63.6	24.3	12.1	0.070	1.000	14.370	
8325	T	*M. cerifera*	1	0.0726	34.33	81	4	15	0.096	2.107	21.918	
8350	R	*M. cerifera*	0	0.0511	67.19	69.9	11.7	18.4	0.097	1.468	15.198	
8350	T	*M. cerifera*	1	0.0505	59.91	59.6	24	16.4	0.151	2.362	15.683	
8375	T	*M. cerifera*	1	0.0413	60.08	69.9	15.7	14.4	0.123	1.430	11.651	
8375	R	*M. cerifera*	0	0.0681	52.13	47.6	26	26.4	0.133	1.906	14.334	
8400	R	*M. cerifera*	0	0.0541	45.15	60.4	13.7	25.9	0.118	1.787	15.197	
8425	R	*M. cerifera*	0	0.0884	53.89	64.7	12.3	23	0.190	2.786	14.687	
8450	T	*M. cerifera*	1	0.1182	48.8	64.7	20	15.3	0.163	1.991	12.246	
8450	R	*M. cerifera*	0	0.0877	43.38	65.3	20.3	14.4	0.133	1.906	14.334	
8475	T	*M. cerifera*	1	0.054	51.93	75.3	11.7	13	0.069	0.871	12.709	
8475	R	*M. cerifera*	0	0.0404	42.9	73.3	18	8.7	0.048	0.683	14.099	
8500	T	*M. cerifera*	1	0.1515	23.67	57.6	20	22.4	0.132	1.860	14.039	
8500	R	*M. cerifera*	0	0.1688	20.34	69.6	14.3	16.1	0.146	2.516	17.207	

Appendix A. Raw data from observational study, continued

8525	R	*M. cerifera*	0	0.072	30.61	82.7	5.7	11.6	0.114	1.862	16.287
8525	T	*M. cerifera*	1	0.135	22.88	65.8	8.3	25.9	0.145	2.481	17.077
8550	R	*M. cerifera*	0	0.1347	28.52	70.1	8.6	21.3	0.222	4.004	18.053
8575	R	*M. cerifera*	0	0.1697	27.74	63.3	18	18.7	0.149	3.519	23.540
8600	R	*M. cerifera*	0	0.1765	32.6	64.1	14.6	21.3	0.313	5.385	17.226
8625	T	*M. cerifera*	1	0.151	30.94	52.7	20.6	26.7	0.257	4.534	17.662
8625	R	*M. cerifera*	0	0.1483	26.19	63.3	12	24.7	0.200	3.562	17.821
8650	R	*M. cerifera*	0	0.1605	20.07	76.1	8.3	15.6	0.222	4.231	19.099
8675	R	*M. cerifera*	0	0.223	19.62	47	35.8	17.2	0.299	5.128	17.151
8675	T	*M. cerifera*	1	0.207	27.03	57.3	10.3	32.4	0.301	5.730	19.017
8700	R	*M. cerifera*	0	0.0721	18.48	79.5	6.8	13.7	0.206	3.890	18.925
8725	R	*M. cerifera*	0	0.303	72.66	69.7	2.4	27.9	0.370	6.395	17.280
8750	R	*M. cerifera*	0	0.2182	72.44	56.1	12.6	31.3	0.222	3.859	17.397
8775	T	*M. cerifera*	1	0.1629	25.31	68.7	14.5	16.8	0.288	4.513	15.656
8775	R	*M. cerifera*	0	0.19	14.74	7	46.4	46.6	0.438	8.230	18.800
8800	R	*M. cerifera*	0	0.1638	34.75	26.3	20.9	52.8	0.265	4.008	15.152
8825	R	*M. cerifera*	0	0.115	36.6	51.3	18	30.7	0.139	2.599	18.730
8850	R	*M. cerifera*	0	0.115	37.63	41	22.6	36.4	0.292	4.033	13.816
8850	T	*M. cerifera*	1	0.115	31.77	51.9	21.6	26.5	0.152	2.587	17.004
8875	T	*M. cerifera*	1	0.0914	18.38	50.7	24.9	24.4	0.142	2.205	15.494
8875	R	*M. cerifera*	0	0.1173	23.32	55	22	23	0.183	2.993	16.392
8900	T	*M. cerifera*	1	0.112	31.45	56.2	20.3	23.5	0.103	1.287	12.550
8900	R	*M. cerifera*	0	0.1055	24.93	67	18	15	0.139	2.562	18.431
8925	T	*M. cerifera*	1	0.0956	22.31	78.5	13.5	8	0.171	3.697	21.663
8925	R	*M. cerifera*	0	0.088	35.48	79.2	8.3	12.5	0.108	3.293	30.380
8950	T	*M. cerifera*	1	0.0724	30.13	60.2	14	25.8	0.167	3.137	18.791
8950	R	*M. cerifera*	0	0.0844	21.03	55.9	20.3	23.8	0.172	4.089	23.827
8975	T	*M. cerifera*	1	0.0845	22.72	58.2	16.6	25.2	0.183	3.445	18.777
8975	R	*M. cerifera*	0	0.1359	22.16	50.9	26.4	22.7	0.285	5.363	18.828
9000	T	*M. cerifera*	1	0.0847	22.1	53.3	20.3	26.4	0.244	4.533	18.560
9000	R	*M. cerifera*	0	0.0903	23.51	48.6	24.6	26.8	0.102	2.218	21.688
9025	T	*M. cerifera*	1	0.097	14.49	50.2	20.3	29.5	0.165	3.052	18.495
9025	R	*M. cerifera*	0	0.111	21.47	58.5	18.3	23.2	0.298	6.009	20.186
9050	T	*M. cerifera*	1	0.093	18.84	75.2	12.3	12.5	0.101	1.866	18.442
9050	R	*M. cerifera*	0	0.0855	13.13	61.3	8.6	30.1	0.145	2.958	20.410
9075	T	*M. cerifera*	1	0.0614	25.92	69.5	16.6	13.9	0.151	2.536	16.819

Appendix A. Raw data from observational study, continued

9075	R	*M. cerifera*	0	0.0732	25.52	81.3	6.3	12.4	0.181	3.206	17.689
9100	T	*M. cerifera*	1	0.171	22	60	17.3	22.7	0.113	1.740	15.424
9100	R	*M. cerifera*	0	0.2022	9.16	69.2	11.2	19.6	0.251	4.753	18.925
9125	R	*M. cerifera*	0	0.0959	9.69	52.7	18.6	28.7	0.170	2.880	16.910
9125	T	*M. cerifera*	1	0.0498	30.72	69.3	9.2	21.5	0.085	1.557	18.275
9150	T	*M. cerifera*	1	0.0733	53.31	55.9	28.9	15.2	0.110	1.836	16.642
9150	R	*M. cerifera*	0	0.0493	52.46	75.5	13.2	11.3	0.107	1.870	17.411
9175	T	*M. cerifera*	1	0.0438	41.78	62.5	16.6	20.9	0.140	1.627	11.620
9175	R	*M. cerifera*	0	0.0435	43.66	65.9	18.3	15.8	0.084	1.313	15.631
9200	T	*M. cerifera*	1	0.031	51.47	62.2	16.3	21.5	0.112	1.355	12.064
9200	R	*M. cerifera*	0	0.0299	49.2	71.8	14.9	13.3	0.087	1.453	16.735
9225	R	*M. cerifera*	0	0.0718	39.67	61	16.3	22.7	0.141	1.750	12.434
9225	T	*M. cerifera*	1	0.068	41.25	75.5	16.9	7.6	0.072	1.284	17.926
9250	T	*M. cerifera*	1	0.057	30.56	50.6	17.6	31.8	0.181	2.212	12.252
9250	R	*M. cerifera*	0	0.0756	28.23	67.2	9.2	23.6	0.150	2.980	19.823
9275	T	*M. cerifera*	1	0.101	26.02	66.4	14.6	19	0.115	1.848	16.051
9275	R	*M. cerifera*	0	0.1012	31.31	61	16.3	22.7	0.150	4.022	26.825
9300	R	*M. cerifera*	0	0.108	13.04	62.7	14.6	22.7	0.141	2.344	16.650
9300	T	*M. cerifera*	1	0.0867	23.75	61.9	16	22.1	0.107	1.903	17.827
9325	T	*M. cerifera*	1	0.2053	19.76	59	18	23	0.079	1.150	14.502
9325	R	*M. cerifera*	0	0.1581	14.16	55.6	20.3	24.1	0.071	1.044	14.667
9350	T	*M. cerifera*	1	0.087	14.15	61.3	16	22.7	0.065	0.739	11.453
9350	R	*M. cerifera*	0	0.0655	24.41	62.6	19.3	18.1	0.171	2.599	15.168
9375	R	*M. cerifera*	0	0.0825	16.24	58.4	28.3	13.3	0.075	1.145	15.196
9375	T	*M. cerifera*	1	0.0845	10.88	81.3	10.3	8.4	0.152	2.822	18.595
9400	R	*M. cerifera*	0	0.0952	10.56	62.1	6.6	31.3	0.123	2.198	17.842
9400	T	*M. cerifera*	1	0.1213	17.53	72.9	19.8	7.3	0.202	3.865	19.111
9425	R	*M. cerifera*	0	0.1595	9.14	71.3	16	12.7	0.073	1.042	14.313
9425	T	*M. cerifera*	1	0.099	11.6	75.3	10	14.7	0.150	2.301	15.328
9450	R	*M. cerifera*	0	0.0913	28.23	78.1	8.9	13	0.135	2.179	16.087
9450	T	*M. cerifera*	1	0.135	30.56	63.9	22.3	13.8	0.256	4.199	16.402
9475	R	*M. cerifera*	0	0.074	31.31	65.3	14.3	20.4	0.200	2.090	10.438
9475	T	*M. cerifera*	1	0.0685	26.02	74.5	8.6	16.9	0.152	2.822	18.595
9500	T	*M. cerifera*	1	0.0905	13.46	77.5	6.9	15.6	0.218	3.143	14.424
9500	R	*M. cerifera*	0	0.135	12.25	79.6	12	8.4	0.066	1.559	23.722
9525	R	*M. cerifera*	0	0.1056	16.13	68.5	10.3	21.2	0.257	4.048	15.755

Appendix A. Raw data from observational study, continued

9525	T	*M. cerifera*	1	0.0959	24.2	63.8	15	21.2	0.200	3.530	17.609
9550	R	*M. cerifera*	0	0.086	16.58	60.9	19.3	19.8	0.260	4.991	19.161
9575	R	*M. cerifera*	0	0.1358	24.26	47	31.6	21.4	0.326	5.628	17.243
9600	T	*M. cerifera*	1	0.1673	17.32	41.2	32.4	26.4	0.486	8.120	16.720
9600	R	*M. cerifera*	0	0.084	27.25	69.2	13.2	17.6	0.208	3.947	18.985
9625	R	*M. cerifera*	0	0.0983	15.26	63.6	16.6	19.8	0.178	3.883	21.844
9625	T	*M. cerifera*	1	0.1111	12.64	54.7	18.3	27	0.461	10.236	22.194
9650	T	*M. cerifera*	1	0.118	14.35	34.4	43	22.6	0.289	4.775	16.498
9650	R	*M. cerifera*	0	0.0962	12.04	57.3	16.3	26.4	0.201	4.176	20.803
9675	R	*M. cerifera*	0	0.125	8.94	65.5	13.7	20.8	0.378	6.589	17.428
9675	T	*M. cerifera*	1	0.102	13.21	58.2	16.6	25.2	0.203	4.190	20.658
9700	T	*M. cerifera*	1	0.1	10.04	65.9	4	30.1	0.276	4.685	16.958
9700	R	*M. cerifera*	0	0.12	18.15	11	58.4	30.6	0.366	6.633	18.112
9725	R	*M. cerifera*	0	0.0845	10.87	61.3	16.6	22.1	0.222	3.743	16.830
9725	T	*M. cerifera*	1	0.0935	13.13	67	12.3	20.7	0.227	4.312	18.962
9750	R	*M. cerifera*	0	0.0829	11.87	43.6	22.3	34.1	0.259	4.329	16.702
9750	T	*M. cerifera*	1	0.079	13.42	47.9	24	28.1	0.162	3.017	18.648
9775	R	*M. cerifera*	0	0.082	22.14	48.2	20.3	31.5	0.192	3.044	15.848
9775	T	*M. cerifera*	1	0.145	20.4	56.5	15.6	27.9	0.158	3.029	19.143
9800	R	*M. cerifera*	0	0.1081	29.32	60.7	12	27.3	0.228	3.361	14.770
9800	T	*M. cerifera*	1	0.0631	15.36	43	26.3	30.7	0.130	2.690	20.743
9825	R	*M. cerifera*	0	0.1041	30.47	48.1	20	31.9	0.189	3.015	15.958
9825	T	*M. cerifera*	1	0.0823	26.91	33.3	28.3	38.4	0.285	4.634	16.283
9850	T	*M. cerifera*	1	0.066	16.71	46.5	20.6	32.9	0.169	3.032	17.953
9850	R	*M. cerifera*	0	0.0845	29.94	66.1	12.8	21.1	0.206	3.814	18.526
9875	R	*M. cerifera*	0	0.1145	21.32	29	26.3	44.7	0.319	5.466	17.130
9900	R	*M. cerifera*	0	0.0981	26.04	12.5	32.3	55.2	0.232	4.045	17.429
9900	T	*M. cerifera*	1	0.1492	23.02	31.3	30.3	38.4	0.324	5.974	18.422
9925	R	*M. cerifera*	0	0.19	18.57	49.6	16.3	34.1	0.392	6.610	16.879
9925	T	*M. cerifera*	1	0.1523	21.44	88	8	4	0.223	4.799	21.501
9950	R	*M. cerifera*	0	0.0988	15.88	38.5	25.3	36.2	0.214	4.588	21.438
9975	R	*M. cerifera*	0	0.0953	17.93	44.5	32.5	23	0.225	3.610	16.021
9975	T	*M. cerifera*	1	0.0583	18.95	47.8	20.9	31.3	0.141	2.896	20.509
10000	T	*M. cerifera*	1	0.0879	19.81	39.8	26.3	33.9	0.120	1.590	13.289
10000	R	*M. cerifera*	0	0.06	52.3	36.5	28.3	35.2	0.100	1.828	18.267
10025	R	*M. cerifera*	0	0.0305	28.24	46.7	20.3	33	0.211	3.027	14.382

Appendix A. Raw data from observational study, continued

10025	T	*M. cerifera*	1	0.0629	19.01	51.3	20.3	28.4	0.211	3.027	14.382
10075	R	*M. cerifera*	0	0.04	33.01	53.1	20.6	26.3	0.247	5.176	20.971
10100	R	*M. cerifera*	0	0.0528	27.8	17.2	50.4	32.4	0.289	3.897	13.495
10125	R	*M. cerifera*	0	0.0207	21.43	52.38	18.42	29.2	0.089	1.343	15.163
10125	T	*M. cerifera*	1	0.036	10.87	57.3	22.6	20.1	0.173	3.260	18.789
10150	T	*M. cerifera*	1	0.0434	22.27	62.7	14.6	22.7	0.185	2.909	15.753
10150	R	*M. cerifera*	0	0.0553	21.61	29.8	46.4	23.8	0.417	7.319	17.544
10175	R	*M. cerifera*	0	0.0645	21.96	65.6	18	16.4	0.243	3.827	15.767
10175	T	*M. cerifera*	1	0.035	23.43	53.9	20.3	25.8	0.147	2.840	19.316
10200	R	*M. cerifera*	0	0.0403	28.84	43	26.3	30.7	0.227	3.117	13.733
10200	T	*M. cerifera*	1	0.0405	17.48	57.9	21.7	20.4	0.137	2.366	17.327
10225	R	*M. cerifera*	0	0.0522	32.16	48.9	25.3	25.8	0.170	2.776	16.304
10250	R	*M. cerifera*	0	0.036	30.71	47.6	26.3	26.1	0.213	3.222	15.105
10250	T	*M. cerifera*	1	0.034	29.93	42.5	30.3	27.2	0.165	2.757	16.747
10275	R	*M. cerifera*	0	0.0313	30.26	32.5	30.3	37.2	0.151	2.570	16.968
10275	T	*M. cerifera*	1	0.042	32.59	37.6	30.6	31.8	0.140	2.412	17.186
10300	R	*M. cerifera*	0	0.0433	39.19	52.4	20.6	27	0.211	3.061	14.472
10325	R	*M. cerifera*	0	0.045	34.91	34.7	30.6	34.7	0.223	2.589	11.625
10350	R	*M. cerifera*	0	0.0287	52.01	42.5	26.3	31.2	0.117	1.486	12.669
10350	T	*M. cerifera*	1	0.0355	47.08	40.5	31.6	27.9	0.144	1.990	13.807
10375	T	*M. cerifera*	1	0.086	51.32	49.1	28.9	22	0.179	2.530	14.136
10375	R	*M. cerifera*	0	0.066	50.84	53.3	25.7	21	0.189	2.834	14.966
10400	T	*M. cerifera*	1	0.0514	54.72	48.5	24.6	26.9	0.188	2.612	13.885
10400	R	*M. cerifera*	0	0.0785	48.61	36.2	34.3	29.5	0.341	5.620	16.470
10425	T	*M. cerifera*	1	0.0557	55.33	36.2	34.3	29.5	0.175	2.378	13.612
10425	R	*M. cerifera*	0	0.073	67.26	43.4	32	24.6	0.139	1.909	13.765
10450	R	*M. cerifera*	0	0.0466	73.87	50.8	22.72	26.48	0.086	0.962	11.244
10450	T	*M. cerifera*	1	0.0526	66.61	55.3	26.3	18.4	0.158	2.211	14.032
10475	T	*M. cerifera*	1	0.0437	75.71	65.5	19.2	15.3	0.099	1.057	10.637
10475	R	*M. cerifera*	0	0.0455	57.57	42.8	30.6	26.6	0.099	1.228	12.465
10500	T	*M. cerifera*	1	0.0452	70.77	66.7	15.7	17.6	0.145	1.771	12.230
10500	R	*M. cerifera*	0	0.0411	54.45	45.3	30.3	24.4	0.093	1.269	13.571
10525	T	*M. cerifera*	1	0.033	71.65	48.7	22.6	28.7	0.104	1.279	12.319
10525	R	*M. cerifera*	0	0.0485	72.95	62.2	21.4	16.4	0.099	1.362	13.733
10550	R	*M. cerifera*	0	0.044	61.7	57.6	23.7	18.7	0.097	1.259	12.939
10550	T	*M. cerifera*	1	0.0491	74.84	56.8	19.6	23.6	0.121	1.645	13.544

Appendix A. Raw data from observational study, continued

10575	R	*M. cerifera*	0	0.0346	78.04	56.1	24	19.9	0.110	1.476	13.356
10575	T	*M. cerifera*	1	0.044	73.14	61.6	15.1	23.3	0.109	1.512	13.817
10600	R	*M. cerifera*	0	0.0446	82.43	36.5	34.3	29.2	0.126	1.540	12.237
10600	T	*M. cerifera*	1	0.0526	77.06	71	10	19	0.156	1.987	12.714
10625	T	*M. cerifera*	1	0.0646	81.78	61.6	24.6	13.8	0.158	2.142	13.570
10625	R	*M. cerifera*	0	0.0493	71.82	71.5	16	12.5	0.130	1.871	14.358
10650	R	*M. cerifera*	0	0.024	82.84	70.7	6	23.3	0.146	1.476	10.121
10650	T	*M. cerifera*	1	0.0269	75.2	61.4	14	24.6	0.138	1.941	14.107
10675	T	*M. cerifera*	1	0.0206	76.93	72.7	10	17.3	0.117	1.309	11.225
10675	R	*M. cerifera*	0	0.0258	66.14	74.7	4	21.3	0.102	1.385	13.600
10700	R	*M. cerifera*	0	0.031	76.02	58.2	16.6	25.2	0.092	1.142	12.463
10700	T	*M. cerifera*	1	0.0204	74.14	67.9	6.8	25.3	0.092	1.244	13.530
10725	R	*M. cerifera*	0	0.0228	66.72	38.8	37.3	23.9	0.150	2.161	14.384
10725	T	*M. cerifera*	1	0.0236	56.94	29.6	36.6	33.8	0.186	2.796	15.053
10750	R	*M. cerifera*	0	0.0344	62.6	75.5	7.7	16.8	0.129	1.839	14.299
10750	T	*M. cerifera*	1	0.0368	62.76	41.3	32.6	26.1	0.240	4.025	16.742
10775	R	*M. cerifera*	0	0.023	59.38	30.8	30.6	38.6	0.160	2.164	13.525
10775	T	*M. cerifera*	1	0.0302	72.88	36.8	24.6	38.6	0.220	3.141	14.258
10800	T	*M. cerifera*	1	0.0288	76.39	30.2	36.6	33.2	0.171	1.989	11.646
10800	R	*M. cerifera*	0	0.0436	75.6	34.7	30.3	35	0.219	3.114	14.206
10825	T	*M. cerifera*	1	0.0253	79.57	34.5	32.6	32.9	0.146	1.828	12.558
10825	R	*M. cerifera*	0	0.0291	82.53	43.2	26.3	30.5	0.185	2.380	12.843
10850	R	*M. cerifera*	0	0.039	78.25	26.1	37.3	36.6	0.141	1.808	12.855
10850	T	*M. cerifera*	1	0.0331	81.63	38.2	32.3	29.5	0.137	1.828	13.306
10875	R	*M. cerifera*	0	0.0392	77.51	36.8	30.6	32.6	0.218	2.423	11.116
10875	T	*M. cerifera*	1	0.0383	82.94	37.3	32.6	30.1	0.162	2.204	13.648
10900	R	*M. cerifera*	0	0.04	83.19	35	34.6	30.4	0.285	3.128	10.972
10900	T	*M. cerifera*	1	0.0434	85.31	39.3	22.3	38.4	0.224	2.924	13.077
10925	R	*M. cerifera*	0	0.0394	87.64	62.5	18	19.5	0.113	1.812	16.084
10950	R	*M. cerifera*	0	0.037	88.3	76.1	10	13.9	0.104	1.329	12.814
10950	T	*M. cerifera*	1	0.0516	88.54	67.6	20.3	12.1	0.156	2.283	14.676
10975	R	*M. cerifera*	0	0.0307	87.67	61.3	18.3	20.4	0.124	1.369	11.056
10975	T	*M. cerifera*	1	0.035	88.48	61.9	18.6	19.5	0.080	1.041	12.982
11000	T	*M. cerifera*	1	0.0256	87.85	65.8	10.6	23.6	0.181	2.625	14.482
11000	R	*M. cerifera*	0	0.029	87.68	50.1	24.3	25.6	0.155	2.323	15.020
11025	R	*M. cerifera*	0	0.0232	86.4	83.3	10.3	6.4	0.040	0.511	12.761

Appendix A. Raw data from observational study, continued

11025	T	*M. cerifera*	1	0.0477	87.43	68.5	16.3	15.2	0.114	1.590	13.938
11050	T	*M. cerifera*	1	0.0311	83.66	62.5	20	17.5	0.113	0.955	8.416
11050	R	*M. cerifera*	0	0.0262	85.57	82.2	10	7.8	0.115	1.760	15.250
11075	R	*M. cerifera*	0	0.019	85.62	73.3	4.6	22.1	0.061	0.503	8.178
11075	T	*M. cerifera*	1	0.0278	80.8	63.6	14.6	21.8	0.093	0.908	9.774
11100	R	*M. cerifera*	0	0.0227	87.24	83.2	11.2	5.6	0.069	0.581	8.399
11100	T	*M. cerifera*	1	0.0252	85.96	56.8	28.6	14.6	0.148	2.182	14.705
11125	T	*M. cerifera*	1	0.0154	86.06	36.5	36.3	27.2	0.176	2.317	13.160
11125	R	*M. cerifera*	0	0.0306	85.4	44.5	28.9	26.6	0.147	2.111	14.316
11150	T	*M. cerifera*	1	0.0206	84.51	42.8	32.6	24.6	0.114	1.167	10.243
11150	R	*M. cerifera*	0	0.03	84.44	51.3	29.3	19.4	0.189	3.501	18.529
11175	T	*M. cerifera*	1	0.1913	81.44	54.2	19.3	26.5	0.161	1.883	11.660
11175	R	*M. cerifera*	0	0.0256	85.92	61.5	18.9	19.6	0.144	1.762	12.241
11200	T	*M. cerifera*	1	0.0253	81.94	31.9	34.6	33.5	0.171	2.227	13.049
11200	R	*M. cerifera*	0	0.0295	85.75	48.4	28.9	22.7	0.182	2.480	13.643
11225	R	*M. cerifera*	0	0.06	81.15	55.3	26.3	18.4	0.158	1.960	12.395
11225	T	*M. cerifera*	1	0.0717	85.15	47.3	30	22.7	0.187	2.469	13.201
11250	R	*M. cerifera*	0	0.0716	75.79	23.1	40.6	36.3	0.197	2.841	14.446
11250	T	*M. cerifera*	1	0.0686	83.68	35.1	32.3	32.6	0.224	3.490	15.569
11275	T	*M. cerifera*	1	0.0628	77.18	47.5	22.6	29.9	0.203	3.172	15.662
11275	R	*M. cerifera*	0	0.1037	74.05	65.8	14.9	19.3	0.147	2.854	19.361
11300	R	*M. cerifera*	0	0.0814	57.24	33.5	34.3	32.2	0.320	4.026	12.584
11300	T	*M. cerifera*	1	0.085	55.71	28.5	38.9	32.6	0.290	4.329	14.921
0	R	*T. sebifera*	0	6.66	98.15	73.6	10.6	15.8	0.201	3.136	15.616
100	R	*T. sebifera*	0	7.53	98.15	30.4	43	26.6	0.386	4.648	12.035
200	R	*T. sebifera*	0	9.4	98.15	61.8	18.3	19.9	0.373	5.527	14.819
300	R	*T. sebifera*	0	9.03	98.15	59.1	30.9	10	0.459	5.935	12.939
400	R	*T. sebifera*	0	7.4	98.15	13.8	43.6	42.6	0.291	4.170	14.353
500	R	*T. sebifera*	0	9.28	98.15	56.8	16.9	26.3	0.173	2.682	15.472
600	R	*T. sebifera*	0	8.49	98.15	57.1	18.3	24.6	0.324	4.831	14.893
700	R	*T. sebifera*	0	3.567	98.15	67	14	19	0.047	0.424	8.967
800	R	*T. sebifera*	0	8.59	98.15	34.4	35	30.6	0.253	4.045	15.992
900	R	*T. sebifera*	0	10.88	98.15	35	38.4	26.6	0.318	4.948	15.583
1000	R	*T. sebifera*	0	7.38	98.15	69.5	18.6	11.9	0.280	4.108	14.654
1100	R	*T. sebifera*	0	10.1	98.15	27	39	34	0.289	3.897	13.495
1200	R	*T. sebifera*	0	11.53	98.15	.	.	.	0.451	6.415	14.211

Appendix A. Raw data from observational study, continued

1300	R	*T. sebifera*	0	8.94	98.15	74.4	6.4	19.2	0.467	6.793	14.551
1400	R	*T. sebifera*	0	6.43	98.15	59.5	24.3	16.2	0.215	2.749	12.772
1500	R	*T. sebifera*	0	10	98.15	37	52	11	0.436	5.976	13.693
1600	R	*T. sebifera*	0	8.34	98.15	73.2	12.3	14.5	0.411	6.091	14.814
1700	R	*T. sebifera*	0	7.3	98.15	30.4	51	18.6	0.272	3.751	13.791
1800	R	*T. sebifera*	0	9	98.15	29	49.7	21.3	0.317	4.648	14.678
1900	R	*T. sebifera*	0	8	98.15	43	37	20	0.282	4.293	15.245
2000	R	*T. sebifera*	0	11.7	98.15	32.4	27.2	40.4	0.381	5.262	13.821
2100	R	*T. sebifera*	0	8.75	98.15	67.6	16.3	16.1	0.154	2.002	12.975
2200	R	*T. sebifera*	0	11.63	98.15	13.8	26.4	59.8	0.284	3.950	13.903
2300	R	*T. sebifera*	0	5.7	98.15	39.4	36.3	24.3	0.095	1.060	11.213
2400	R	*T. sebifera*	0	8.725	98.15	67.1	10.3	22.6	0.375	6.057	16.170
2500	R	*T. sebifera*	0	10.28	98.15	65.1	15.4	19.5	0.312	5.125	16.407
2600	R	*T. sebifera*	0	9.76	98.15	59.5	21.2	19.3	0.324	4.831	14.893
2700	R	*T. sebifera*	0	9.66	98.15	70.9	13.2	15.9	0.386	5.829	15.113
2800	R	*T. sebifera*	0	10.5	98.15	67.2	11.5	21.3	0.345	5.384	15.610
2900	R	*T. sebifera*	0	5.76	98.15	59.5	18.9	21.6	0.217	2.588	11.923
3000	R	*T. sebifera*	0	9.19	98.15	39.6	39	21.4	0.389	5.392	13.861
3100	R	*T. sebifera*	0	7.91	98.15	64.1	22	13.9	0.364	5.916	16.244
3200	R	*T. sebifera*	0	8.5	98.15	57.5	20.6	21.9	0.263	3.985	15.147
3300	R	*T. sebifera*	0	9.3	98.15	45.8	19.6	34.6	0.281	4.065	14.476
3400	R	*T. sebifera*	0	7.11	98.15	60.5	31.2	8.3	0.433	6.931	15.991
3500	R	*T. sebifera*	0	7.1	98.15	64.1	18.6	17.3	0.446	6.784	15.200
3600	R	*T. sebifera*	0	9.668	98.15	49.1	40.6	10.3	0.472	6.799	14.396
3700	R	*T. sebifera*	0	8.72	98.15	39	33.2	27.8	0.208	2.956	14.241
3800	R	*T. sebifera*	0	13	98.15	67.5	11.2	21.3	0.418	7.770	18.566
3900	R	*T. sebifera*	0	15.43	98.15	48.9	39.8	11.3	0.464	9.034	19.483
4000	R	*T. sebifera*	0	14.6	98.15	59.5	25.5	15	1.079	11.161	10.342
4100	R	*T. sebifera*	0	18.46	98.15	56.8	29.9	13.3	0.482	8.177	16.957
4200	R	*T. sebifera*	0	16.25	97.87	68.8	19.3	11.9	0.420	9.002	21.415
4300	R	*T. sebifera*	0	6.07	97.69	59.9	18.9	21.2	0.151	3.122	20.669
4400	R	*T. sebifera*	0	3.93	90.81	51.6	28	20.4	0.079	1.297	16.380
4500	R	*T. sebifera*	0	3.9	89.67	39.2	42.7	18.1	0.060	0.240	4.003
4600	R	*T. sebifera*	0	4.72	88.53	37	35.2	27.8	0.148	1.914	12.927
4625	R	*T. sebifera*	0	3.891	85.58	52.2	22.3	25.5	0.071	1.055	14.817
4650	R	*T. sebifera*	0	0.892	73.73	57.3	26.3	16.4	0.068	0.899	13.266

Appendix A. Raw data from observational study, continued

4675	R	*T. sebifera*	0	0.4494	60.46	81.5	7.2	11.3	0.068	1.037	15.159
4700	R	*T. sebifera*	0	0.463	59.45	63.3	14.3	22.4	0.091	1.229	13.512
4725	R	*T. sebifera*	0	0.2	58.7	57.9	24.6	17.5	0.106	0.957	9.032
4750	R	*T. sebifera*	0	0.3425	43.58	35.8	36	28.2	0.083	0.702	8.454
4775	R	*T. sebifera*	0	0.2601	40.31	59	20.3	20.7	0.149	2.775	18.569
4800	R	*T. sebifera*	0	0.3266	40.16	76.4	11.8	11.8	0.053	0.694	13.180
4825	R	*T. sebifera*	0	2.652	48.08	49	22	29	0.060	0.967	16.123
4850	R	*T. sebifera*	0	1.354	71.21	51	21.7	27.3	0.127	1.887	14.864
4875	R	*T. sebifera*	0	1.5774	79.52	54.4	28.5	17.1	0.114	1.741	15.319
4900	R	*T. sebifera*	0	3.735	85.12	10.2	68	21.8	0.181	3.492	19.271
4925	R	*T. sebifera*	0	4.35	84.05	64.7	17.8	17.5	0.379	6.877	18.142
4950	R	*T. sebifera*	0	4.75	81.5	77.3	13.5	9.2	0.556	8.788	15.808
4975	R	*T. sebifera*	0	2.28	83.92	52.7	19.7	27.6	0.159	3.044	19.125
5000	R	*T. sebifera*	0	1.012	84.23	79.5	3.2	17.3	0.153	2.412	15.736
5025	R	*T. sebifera*	0	2.025	85.44	59	20.9	20.1	0.103	1.951	18.928
5050	R	*T. sebifera*	0	1.4	79.42	56.1	20.6	23.3	0.174	2.207	12.677
5075	R	*T. sebifera*	0	1.444	71.6	53	20.9	26.1	0.104	1.477	14.220
5100	R	*T. sebifera*	0	0.4307	48.14	61.6	14.3	24.1	0.105	1.699	16.182
5125	R	*T. sebifera*	0	0.332	48.52	59	14	27	0.123	1.952	15.829
5150	R	*T. sebifera*	0	0.547	46.11	59.6	18.3	22.1	0.124	1.595	12.843
5175	R	*T. sebifera*	0	0.4485	42.44	68.4	16.6	15	0.076	1.050	13.791
5200	T	*T. sebifera*	1	0.5	44.2	63.9	17.7	18.4	0.074	0.881	11.876
5200	R	*T. sebifera*	0	0.4485	40.58	65	16.3	18.7	0.062	1.059	17.190
5225	R	*T. sebifera*	0	1.601	70.58	45	18	37	0.244	3.834	15.703
5250	R	*T. sebifera*	0	1.788	70.47	59.6	18.3	22.1	0.243	2.772	11.408
5275	T	*T. sebifera*	1	0.2109	40.13	61.6	14.3	24.1	0.150	1.555	10.344
5275	R	*T. sebifera*	0	0.2601	39.15	65	12.3	22.7	0.130	2.107	16.204
5300	R	*T. sebifera*	0	0.3201	43.3	57.6	18.3	24.1	0.103	1.755	17.083
5325	R	*T. sebifera*	0	0.18	35.6	66.1	12	21.9	0.078	1.433	18.342
5350	T	*T. sebifera*	1	0.1349	33.65	67.5	7.2	25.3	0.149	1.495	10.051
5350	R	*T. sebifera*	0	0.1334	33.62	62.7	16.6	20.7	0.092	1.584	17.249
5375	T	*T. sebifera*	1	0.1785	39.82	67.6	12	20.4	0.087	1.084	12.404
5375	R	*T. sebifera*	0	0.458	43.7	63.9	14.6	21.5	0.078	1.493	19.139
5400	R	*T. sebifera*	0	0.176	28.58	67.5	17	15.5	0.106	1.381	12.969
5425	R	*T. sebifera*	0	0.1485	58.85	56.7	16.9	26.4	0.095	1.270	13.383
5450	R	*T. sebifera*	0	0.1672	63.7	65.4	20.3	14.3	0.129	1.665	12.887

Appendix A. Raw data from observational study, continued

5475	R	*T. sebifera*	0	0.166	70.8	6	35.4	58.6	0.092	1.197	13.039	
5500	R	*T. sebifera*	0	0.3688	60.92	52.4	16	31.6	0.133	1.420	10.686	
5525	R	*T. sebifera*	0	0.3747	38.75	61.3	10.6	28.1	0.106	1.742	16.488	
5550	R	*T. sebifera*	0	0.1422	46.2	57.9	16.6	25.5	0.101	1.483	14.649	
5575	R	*T. sebifera*	0	0.15	54.43	65.5	14.6	19.9	0.115	1.678	14.581	
5600	R	*T. sebifera*	0	0.1679	58.72	75.5	8	16.5	0.130	2.139	16.404	
5625	R	*T. sebifera*	0	0.2068	56.13	63.9	10.6	25.5	0.143	1.713	11.971	
5650	R	*T. sebifera*	0	0.2255	66.55	65.9	14.3	19.8	0.102	1.703	16.733	
5675	R	*T. sebifera*	0	0.129	66.96	65.3	12.3	22.4	0.101	1.391	13.779	
5700	R	*T. sebifera*	0	0.2276	65.18	60.7	16.3	23	0.106	1.293	12.170	
5700	T	*T. sebifera*	1	0.2067	61.87	64.4	12.6	23	0.136	1.940	14.277	
5725	R	*T. sebifera*	0	0.21	56.41	60.1	14	25.9	0.100	1.646	16.533	
5750	R	*T. sebifera*	0	0.285	63.33	61.5	15.2	23.3	0.093	1.456	15.726	
5775	R	*T. sebifera*	0	0.2423	44.92	64.7	12.3	23	0.106	1.565	14.828	
5800	T	*T. sebifera*	1	0.181	28.15	63.3	16.6	20.1	0.072	0.965	13.412	
5800	R	*T. sebifera*	0	0.4027	24.3	63.3	16.6	20.1	0.091	1.594	17.593	
5825	R	*T. sebifera*	0	0.51	33.81	64.4	14.3	21.3	0.100	1.709	17.137	
5825	T	*T. sebifera*	1	0.693	35.97	65.2	19	15.8	0.126	2.331	18.432	
5850	T	*T. sebifera*	1	0.427	50.09	61.5	11.2	27.3	0.327	5.332	16.323	
5850	R	*T. sebifera*	0	1.022	52.6	6.4	55	38.6	0.247	4.514	18.267	
5875	T	*T. sebifera*	1	0.3721	44.3	61.5	17.2	21.3	0.081	1.525	18.913	
5875	R	*T. sebifera*	0	0.4	43.63	65.9	17	17.1	0.155	3.460	22.365	
5900	T	*T. sebifera*	1	0.373	43.59	62.4	6.3	31.3	0.115	2.477	21.515	
5900	R	*T. sebifera*	0	0.4885	37.75	52.7	22.6	24.7	0.085	1.918	22.645	
5925	T	*T. sebifera*	1	0.3721	42.4	54.1	20	25.9	0.156	2.939	18.801	
5925	R	*T. sebifera*	0	0.2638	43.77	54.7	22.3	23	0.302	5.779	19.113	
5950	R	*T. sebifera*	0	0.383	36.58	65.9	19.7	14.4	0.211	3.764	17.875	
5975	T	*T. sebifera*	1	0.638	35.75	56.4	19.2	24.4	0.155	3.005	19.331	
5975	R	*T. sebifera*	0	0.95	58.57	42.7	22.6	34.7	0.237	4.621	19.496	
6000	R	*T. sebifera*	0	0.2055	43.54	63.3	23.5	13.2	0.164	3.119	18.976	
6000	T	*T. sebifera*	1	0.38	42.16	51.3	22.6	26.1	0.112	3.101	27.714	
6025	T	*T. sebifera*	1	0.13	49.66	66.1	10.6	23.3	0.168	1.808	10.729	
6025	R	*T. sebifera*	0	0.166	44.25	51	22.3	26.7	0.196	3.024	15.453	
6050	R	*T. sebifera*	0	0.2883	54.92	50.7	20.9	28.4	0.186	2.846	15.286	
6050	T	*T. sebifera*	1	0.43	45.85	54.7	15.7	29.6	0.096	1.770	18.471	
6075	R	*T. sebifera*	0	0.2682	58.48	63.3	16.9	19.8	0.171	2.937	17.131	

Appendix A. Raw data from observational study, continued

6100	T	*T. sebifera*	1	0.167	58.88	41.3	26.3	32.4	0.214	1.939	9.054
6100	R	*T. sebifera*	0	0.25	57.08	28.6	35.3	36.1	0.141	2.109	14.969
6125	T	*T. sebifera*	1	0.1227	56.26	43.3	30.36	26.34	0.148	1.658	11.187
6125	R	*T. sebifera*	0	0.3457	52.6	40.2	28.3	31.5	0.148	2.560	17.273
6150	T	*T. sebifera*	1	0.445	35.29	57.3	20	22.7	0.096	1.256	13.110
6150	R	*T. sebifera*	0	0.48	38.27	51.9	24.6	23.5	0.064	0.871	13.596
6175	R	*T. sebifera*	0	0.287	23.89	56.1	16.9	27	0.132	2.041	15.442
6200	R	*T. sebifera*	0	0.1182	47.55	37.9	32.6	29.5	0.091	1.160	12.728
6200	T	*T. sebifera*	1	0.445	45.34	44.1	29.2	26.7	0.106	1.824	17.138
6225	R	*T. sebifera*	0	0.238	37.33	42.4	35.5	22.1	0.072	0.962	13.401
6225	T	*T. sebifera*	1	0.387	34.98	52.7	18	29.3	0.166	2.491	14.988
6250	R	*T. sebifera*	0	0.2337	35.92	50.7	26.6	22.7	0.088	1.361	15.546
6275	T	*T. sebifera*	1	0.1841	33	43.3	38.6	18.1	0.103	1.282	12.428
6275	R	*T. sebifera*	0	0.2557	34.76	63.8	22.3	13.9	0.121	1.803	14.960
6300	R	*T. sebifera*	0	0.158	32.99	61.3	20	18.7	0.095	1.068	11.232
6325	R	*T. sebifera*	0	0.091	42.5	64.7	12.6	22.7	0.147	2.045	13.955
6350	R	*T. sebifera*	0	0.1576	49.81	55.8	14.9	29.3	0.178	2.376	13.379
6375	R	*T. sebifera*	0	0.2454	49.58	31.3	36.3	32.4	0.159	2.341	14.765
6400	R	*T. sebifera*	0	0.1919	51.6	15.3	48	36.7	0.185	2.569	13.887
6425	R	*T. sebifera*	0	0.1511	50.67	39.5	26.6	33.9	0.147	2.009	13.661
6450	R	*T. sebifera*	0	0.3422	44.39	41	28.3	30.7	0.162	2.440	15.055
6475	R	*T. sebifera*	0	0.2442	34.34	50.1	26	23.9	0.152	2.267	14.933
6500	R	*T. sebifera*	0	0.0902	33.79	50.7	18	31.3	0.128	2.176	16.996
6525	R	*T. sebifera*	0	0.1511	22.51	53.5	19.2	27.3	0.122	2.243	18.387
6550	R	*T. sebifera*	0	0.1639	24.73	53	20.3	26.7	0.117	2.069	17.749
6575	R	*T. sebifera*	0	0.2563	32.71	63.6	18.3	18.1	0.080	1.066	13.393
6600	R	*T. sebifera*	0	0.165	27.81	75.3	13.2	11.5	0.084	0.989	11.721
6625	R	*T. sebifera*	0	0.3356	19.7	67.9	22	10.1	0.103	1.512	14.713
6650	R	*T. sebifera*	0	0.181	22.84	19	51	30	0.205	3.380	16.522
6675	R	*T. sebifera*	0	0.1014	32.82	43.6	20	36.4	0.093	1.486	15.936
6700	R	*T. sebifera*	0	0.1347	35.39	62.4	14.9	22.7	0.104	0.940	9.067
6725	R	*T. sebifera*	0	0.1184	35.37	77	4.3	18.7	0.127	2.274	17.859
6750	R	*T. sebifera*	0	0.1147	39.63	65.6	12.3	22.1	0.157	1.890	12.027
6775	R	*T. sebifera*	0	0.22	43.78	74.7	14.9	10.4	0.064	0.734	11.390
6800	R	*T. sebifera*	0	0.2425	40.82	56.7	16.3	27	0.071	1.034	14.602
6825	R	*T. sebifera*	0	0.1897	44.68	54.1	16.9	29	0.081	1.267	15.628

Appendix A. Raw data from observational study, continued

6850	R	*T. sebifera*	0	0.1405	40.71	49.4	16.9	33.7	0.091	1.280	13.989
6875	R	*T. sebifera*	0	0.129	46.23	56.2	20	23.8	0.176	2.872	16.319
6900	R	*T. sebifera*	0	.	49.35	.	.	.	.	.	.
6925	R	*T. sebifera*	0	0.2054	30.22	14.4	58.4	27.2	0.180	3.714	20.666
6950	R	*T. sebifera*	0	0.1992	40.49	70.1	7.2	22.7	0.181	2.198	12.143
6975	R	*T. sebifera*	0	0.133	55.2	57.5	9.2	33.3	0.195	2.527	12.939
7000	R	*T. sebifera*	0	0.3326	49.85	74.1	8.3	17.6	0.200	4.356	21.825
7025	R	*T. sebifera*	0	0.1636	56.56	56.4	18.9	24.7	0.162	3.303	20.339
7050	R	*T. sebifera*	0	0.1195	38.45	71.9	11.7	16.4	0.113	2.075	18.369
7075	R	*T. sebifera*	0	0.1926	44.32	64.4	8.9	26.7	0.172	3.139	18.224
7100	R	*T. sebifera*	0	0.1135	33.72	47	18.3	34.7	0.165	3.255	19.690
7125	R	*T. sebifera*	0	0.1994	47.04	53.3	16.6	30.1	0.184	2.879	15.639
7150	R	*T. sebifera*	0	0.1675	36.65	55	18.9	26.1	0.096	1.699	17.763
7175	R	*T. sebifera*	0	0.1323	36.82	42.2	24.6	33.2	0.153	2.044	13.343
7200	R	*T. sebifera*	0	0.11	47.93	41	28.6	30.4	0.156	2.471	15.850
7225	R	*T. sebifera*	0	0.088	55.98	39	29.6	31.4	0.158	2.289	14.524
7250	R	*T. sebifera*	0	0.1118	50.11	37	32.3	30.7	0.139	2.051	14.719
7275	R	*T. sebifera*	0	0.1181	37.5	36.1	30.6	33.3	0.169	2.232	13.223
7300	R	*T. sebifera*	0	0.115	21.99	59.7	19.9	20.4	0.133	2.063	15.558
8200	T	*T. sebifera*	1	0.1766	39.11	64.1	12.6	23.3	0.101	1.495	14.830
8200	R	*T. sebifera*	0	0.1064	21.69	61	22.3	16.7	0.060	1.083	18.098
8225	T	*T. sebifera*	1	0.27	13.16	49.3	22	28.7	0.078	1.065	13.579
8225	R	*T. sebifera*	0	0.1306	14.91	57	20.6	22.4	0.053	1.129	21.213
8250	R	*T. sebifera*	0	0.0823	14.04	70.1	8.3	21.6	0.093	1.500	16.089
8275	R	*T. sebifera*	0	0.0496	36.55	74.1	12	13.9	0.060	0.826	13.771
8275	T	*T. sebifera*	1	0.0646	39.55	3.6	66.4	30	0.070	1.174	16.791
8300	R	*T. sebifera*	0	0.0504	52.86	74.4	11.5	14.1	0.098	1.482	15.092
8325	R	*T. sebifera*	0	0.0482	58.2	63.6	24.3	12.1	0.070	1.000	14.370
8350	R	*T. sebifera*	0	0.0511	67.19	69.9	11.7	18.4	0.097	1.468	15.198
8375	R	*T. sebifera*	0	0.0681	52.13	47.6	26	26.4	0.133	1.906	14.334
8375	T	*T. sebifera*	1	0.067	56.85	73.8	12.6	13.6	0.146	2.203	15.134
8400	R	*T. sebifera*	0	0.0541	45.15	60.4	13.7	25.9	0.118	1.787	15.197
8425	R	*T. sebifera*	0	0.0884	53.89	64.7	12.3	23	0.190	2.786	14.687
8450	R	*T. sebifera*	0	0.0877	43.38	65.3	20.3	14.4	0.133	1.906	14.334
8475	R	*T. sebifera*	0	0.0404	42.9	73.3	18	8.7	0.048	0.683	14.099
8500	R	*T. sebifera*	0	0.1688	20.34	69.6	14.3	16.1	0.146	2.516	17.207

Appendix A. Raw data from observational study, continued

8525	T	*T. sebifera*	1	0.148	19.63	69.9	18.6	11.5	0.143	2.103	14.700
8525	R	*T. sebifera*	0	0.072	30.61	82.7	5.7	11.6	0.114	1.862	16.287
8550	R	*T. sebifera*	0	0.1347	28.52	70.1	8.6	21.3	0.222	4.004	18.053
8575	T	*T. sebifera*	1	0.0544	26.14	59.5	8.6	31.9	0.145	2.967	20.466
8575	R	*T. sebifera*	0	0.1697	27.74	63.3	18	18.7	0.149	3.519	23.540
8600	R	*T. sebifera*	0	0.1765	32.6	64.1	14.6	21.3	0.313	5.385	17.226
8600	T	*T. sebifera*	1	0.1585	32.27	59.6	22.3	18.1	0.158	3.079	19.498
8625	T	*T. sebifera*	1	0.1581	31.17	61.3	16.6	22.1	0.247	4.168	16.848
8625	R	*T. sebifera*	0	0.1483	26.19	63.3	12	24.7	0.200	3.562	17.821
8650	R	*T. sebifera*	0	0.1605	20.07	76.1	8.3	15.6	0.222	4.231	19.099
8675	R	*T. sebifera*	0	0.1336	19.62	47	35.8	17.2	0.299	5.128	17.151
8675	T	*T. sebifera*	1	0.119	16.56	59.6	28.4	12	0.224	4.137	18.453
8700	R	*T. sebifera*	0	0.0721	18.48	79.5	6.8	13.7	0.206	3.890	18.925
8700	T	*T. sebifera*	1	0.1403	19.51	66.7	16.3	17	0.229	4.415	19.242
8725	R	*T. sebifera*	0	0.303	72.66	69.7	2.4	27.9	0.370	6.395	17.280
8750	R	*T. sebifera*	0	0.2182	72.44	56.1	12.6	31.3	0.222	3.859	17.397
8775	R	*T. sebifera*	0	0.19	14.74	7	46.4	46.6	0.438	8.230	18.800
8800	R	*T. sebifera*	0	0.1638	34.75	26.3	20.9	52.8	0.265	4.008	15.152
8825	R	*T. sebifera*	0	0.115	36.6	51.3	18	30.7	0.139	2.599	18.730
8850	R	*T. sebifera*	0	0.115	37.63	41	22.6	36.4	0.292	4.033	13.816
8875	R	*T. sebifera*	0	0.1173	23.32	55	22	23	0.183	2.993	16.392
8900	T	*T. sebifera*	1	0.154	29.86	81.6	8.6	9.8	0.090	1.136	12.655
8900	R	*T. sebifera*	0	0.1055	24.93	67	18	15	0.139	2.562	18.431
8925	R	*T. sebifera*	0	0.088	35.48	79.2	8.3	12.5	0.108	3.293	30.380
8950	R	*T. sebifera*	0	0.0844	21.03	55.9	20.3	23.8	0.172	4.089	23.827
8975	R	*T. sebifera*	0	0.1359	22.16	50.9	26.4	22.7	0.285	5.363	18.828
9000	R	*T. sebifera*	0	0.0903	23.51	48.6	24.6	26.8	0.102	2.218	21.688
9025	R	*T. sebifera*	0	0.111	21.47	58.5	18.3	23.2	0.298	6.009	20.186
9050	R	*T. sebifera*	0	0.0855	13.13	61.3	8.6	30.1	0.145	2.958	20.410
9075	R	*T. sebifera*	0	0.0732	25.52	81.3	6.3	12.4	0.181	3.206	17.689
9100	R	*T. sebifera*	0	0.2022	9.16	69.2	11.2	19.6	0.251	4.753	18.925
9125	R	*T. sebifera*	0	0.0959	9.69	52.7	18.6	28.7	0.170	2.880	16.910
9150	R	*T. sebifera*	0	0.0493	52.46	75.5	13.2	11.3	0.107	1.870	17.411
9175	R	*T. sebifera*	0	0.0435	43.66	65.9	18.3	15.8	0.084	1.313	15.631
9200	R	*T. sebifera*	0	0.0299	49.2	71.8	14.9	13.3	0.087	1.453	16.735
9225	R	*T. sebifera*	0	0.0718	39.67	61	16.3	22.7	0.141	1.750	12.434

Appendix A. Raw data from observational study, continued

9250	R	*T. sebifera*	0	0.0756	28.23	67.2	9.2	23.6	0.150	2.980	19.823
9275	R	*T. sebifera*	0	0.1012	31.31	61	16.3	22.7	0.150	4.022	26.825
9300	R	*T. sebifera*	0	0.108	13.04	62.7	14.6	22.7	0.141	2.344	16.650
9325	R	*T. sebifera*	0	0.1581	14.16	55.6	20.3	24.1	0.071	1.044	14.667
9350	R	*T. sebifera*	0	0.0655	24.41	62.6	19.3	18.1	0.171	2.599	15.168
9375	R	*T. sebifera*	0	0.0825	16.24	58.4	28.3	13.3	0.075	1.145	15.196
9375	T	*T. sebifera*	1	0.0845	12.71	73.3	15.7	11	0.086	1.337	15.562
9400	R	*T. sebifera*	0	0.0952	10.56	62.1	6.6	31.3	0.123	2.198	17.842
9425	R	*T. sebifera*	0	0.1595	9.14	71.3	16	12.7	0.073	1.042	14.313
9425	T	*T. sebifera*	1	0.1123	11.64	77	14.9	8.1	0.109	1.864	17.163
9450	R	*T. sebifera*	0	0.0913	28.23	78.1	8.9	13	0.135	2.179	16.087
9450	T	*T. sebifera*	1	0.1865	16.26	73	7.3	19.7	0.121	2.103	17.396
9475	R	*T. sebifera*	0	0.074	31.31	65.3	14.3	20.4	0.200	2.090	10.438
9475	T	*T. sebifera*	1	0.0577	30.53	80.7	6.3	13	0.103	1.635	15.870
9500	T	*T. sebifera*	1	0.0648	15.3	44.2	24.4	31.4	0.140	2.204	15.788
9500	R	*T. sebifera*	0	0.135	12.25	79.6	12	8.4	0.066	1.559	23.722
9525	T	*T. sebifera*	1	0.1453	29.46	43	28.4	28.6	0.147	2.130	14.474
9525	R	*T. sebifera*	0	0.1056	16.13	68.5	10.3	21.2	0.257	4.048	15.755
9550	T	*T. sebifera*	1	0.1175	12.35	69	12.6	18.4	0.259	4.324	16.669
9550	R	*T. sebifera*	0	0.086	16.58	60.9	19.3	19.8	0.260	4.991	19.161
9575	R	*T. sebifera*	0	0.1358	24.26	47	31.6	21.4	0.326	5.628	17.243
9575	T	*T. sebifera*	1	0.1401	14.46	40.2	36.4	23.4	0.287	5.518	19.239
9600	T	*T. sebifera*	1	0.2076	20.86	27	42.4	30.6	0.334	6.055	18.119
9600	R	*T. sebifera*	0	0.084	27.25	69.2	13.2	17.6	0.208	3.947	18.985
9625	T	*T. sebifera*	1	0.0939	15.25	27	42.4	30.6	0.252	5.083	20.179
9625	R	*T. sebifera*	0	0.0983	15.26	63.6	16.6	19.8	0.178	3.883	21.844
9650	R	*T. sebifera*	0	0.0962	12.04	57.3	16.3	26.4	0.201	4.176	20.803
9675	T	*T. sebifera*	1	0.078	10.99	55.2	16.3	28.5	0.227	3.944	17.366
9675	R	*T. sebifera*	0	0.125	8.94	65.5	13.7	20.8	0.378	6.589	17.428
9700	T	*T. sebifera*	1	0.0715	10.37	34.7	28.3	37	0.214	3.348	15.669
9700	R	*T. sebifera*	0	0.12	18.15	11	58.4	30.6	0.366	6.633	18.112
9725	R	*T. sebifera*	0	0.0845	10.87	61.3	16.6	22.1	0.222	3.743	16.830
9750	R	*T. sebifera*	0	0.0829	11.87	43.6	22.3	34.1	0.259	4.329	16.702
9775	R	*T. sebifera*	0	0.082	22.14	48.2	20.3	31.5	0.192	3.044	15.848
9800	R	*T. sebifera*	0	0.1081	29.32	60.7	12	27.3	0.228	3.361	14.770
9825	R	*T. sebifera*	0	0.1041	30.47	48.1	20	31.9	0.189	3.015	15.958

Appendix A. Raw data from observational study, continued

9850	R	*T. sebifera*	0	0.0845	29.94	66.1	12.8	21.1	0.206	3.814	18.526
9850	T	*T. sebifera*	1	0.074	17.78	49.1	22.3	28.6	0.208	3.981	19.169
9875	R	*T. sebifera*	0	0.1145	21.32	29	26.3	44.7	0.319	5.466	17.130
9900	R	*T. sebifera*	0	0.0981	26.04	12.5	32.3	55.2	0.232	4.045	17.429
9925	R	*T. sebifera*	0	0.19	18.57	49.6	16.3	34.1	0.392	6.610	16.879
9950	R	*T. sebifera*	0	0.0988	15.88	38.5	25.3	36.2	0.214	4.588	21.438
9975	R	*T. sebifera*	0	0.0953	17.93	44.5	32.5	23	0.225	3.610	16.021
9975	T	*T. sebifera*	1	0.106	18.09	36.8	30.6	32.6	0.229	4.682	20.446
10000	R	*T. sebifera*	0	0.06	52.3	36.5	28.3	35.2	0.100	1.828	18.267
10000	T	*T. sebifera*	1	0.0831	30.17	42.7	22.6	34.7	0.206	4.465	21.640
10025	R	*T. sebifera*	0	0.0305	28.24	46.7	20.3	33	0.211	3.027	14.382
10025	T	*T. sebifera*	1	0.0939	38.34	38.8	27.6	33.6	0.197	3.858	19.626
10075	R	*T. sebifera*	0	0.04	33.01	53.1	20.6	26.3	0.247	5.176	20.971
10075	T	*T. sebifera*	1	0.042	33.93	46.5	26.9	26.6	0.240	6.859	28.551
10100	R	*T. sebifera*	0	0.0528	27.8	17.2	50.4	32.4	0.289	3.897	13.495
10125	R	*T. sebifera*	0	0.0207	21.43	52.38	18.42	29.2	0.089	1.343	15.163
10125	T	*T. sebifera*	1	0.054	20.63	57.7	16.3	26	0.342	6.064	17.706
10150	T	*T. sebifera*	1	0.0444	21.21	50.8	22.3	26.9	0.209	3.402	16.257
10150	R	*T. sebifera*	0	0.0553	21.61	29.8	46.4	23.8	0.417	7.319	17.544
10175	R	*T. sebifera*	0	0.0645	21.96	65.6	18	16.4	0.243	3.827	15.767
10200	R	*T. sebifera*	0	0.0403	28.84	43	26.3	30.7	0.227	3.117	13.733
10225	R	*T. sebifera*	0	0.0522	32.16	48.9	25.3	25.8	0.170	2.776	16.304
10250	R	*T. sebifera*	0	0.036	30.71	47.6	26.3	26.1	0.213	3.222	15.105
10275	R	*T. sebifera*	0	0.0313	30.26	32.5	30.3	37.2	0.151	2.570	16.968
10300	R	*T. sebifera*	0	0.0433	39.19	52.4	20.6	27	0.211	3.061	14.472
10325	R	*T. sebifera*	0	0.045	34.91	34.7	30.6	34.7	0.223	2.589	11.625
10350	R	*T. sebifera*	0	0.0287	52.01	42.5	26.3	31.2	0.117	1.486	12.669
10375	R	*T. sebifera*	0	0.066	50.84	53.3	25.7	21	0.189	2.834	14.966
10400	R	*T. sebifera*	0	0.0785	48.61	36.2	34.3	29.5	0.341	5.620	16.470
10425	R	*T. sebifera*	0	0.073	67.26	43.4	32	24.6	0.139	1.909	13.765
10450	R	*T. sebifera*	0	0.0466	73.87	50.8	22.72	26.48	0.086	0.962	11.244
10475	R	*T. sebifera*	0	0.0455	57.57	42.8	30.6	26.6	0.099	1.228	12.465
10500	R	*T. sebifera*	0	0.0411	54.45	45.3	30.3	24.4	0.093	1.269	13.571
10525	R	*T. sebifera*	0	0.0485	72.95	62.2	21.4	16.4	0.099	1.362	13.733
10550	R	*T. sebifera*	0	0.044	61.7	57.6	23.7	18.7	0.097	1.259	12.939
10575	R	*T. sebifera*	0	0.0346	78.04	56.1	24	19.9	0.110	1.476	13.356

Appendix A. Raw data from observational study, continued

10600	R	*T. sebifera*	0	0.0446	82.43	36.5	34.3	29.2	0.126	1.540	12.237
10625	R	*T. sebifera*	0	0.0493	71.82	71.5	16	12.5	0.130	1.871	14.358
10650	R	*T. sebifera*	0	0.024	82.84	70.7	6	23.3	0.146	1.476	10.121
10675	R	*T. sebifera*	0	0.0258	66.14	74.7	4	21.3	0.102	1.385	13.600
10700	R	*T. sebifera*	0	0.031	76.02	58.2	16.6	25.2	0.092	1.142	12.463
10725	R	*T. sebifera*	0	0.0228	66.72	38.8	37.3	23.9	0.150	2.161	14.384
10750	R	*T. sebifera*	0	0.0344	62.6	75.5	7.7	16.8	0.129	1.839	14.299
10775	R	*T. sebifera*	0	0.023	59.38	30.8	30.6	38.6	0.160	2.164	13.525
10800	R	*T. sebifera*	0	0.0436	75.6	34.7	30.3	35	0.219	3.114	14.206
10825	R	*T. sebifera*	0	0.0291	82.53	43.2	26.3	30.5	0.185	2.380	12.843
10850	R	*T. sebifera*	0	0.039	78.25	26.1	37.3	36.6	0.141	1.808	12.855
10875	R	*T. sebifera*	0	0.0392	77.51	36.8	30.6	32.6	0.218	2.423	11.116
10900	R	*T. sebifera*	0	0.04	83.19	35	34.6	30.4	0.285	3.128	10.972
10925	R	*T. sebifera*	0	0.0394	87.64	62.5	18	19.5	0.113	1.812	16.084
10950	R	*T. sebifera*	0	0.037	88.3	76.1	10	13.9	0.104	1.329	12.814
10975	R	*T. sebifera*	0	0.0307	87.67	61.3	18.3	20.4	0.124	1.369	11.056
11000	R	*T. sebifera*	0	0.029	87.68	50.1	24.3	25.6	0.155	2.323	15.020
11025	R	*T. sebifera*	0	0.0232	86.4	83.3	10.3	6.4	0.040	0.511	12.761
11050	R	*T. sebifera*	0	0.0262	85.57	82.2	10	7.8	0.115	1.760	15.250
11075	R	*T. sebifera*	0	0.019	85.62	73.3	4.6	22.1	0.061	0.503	8.178
11100	R	*T. sebifera*	0	0.0227	87.24	83.2	11.2	5.6	0.069	0.581	8.399
11125	R	*T. sebifera*	0	0.0154	85.4	44.5	28.9	26.6	0.147	2.111	14.316
11150	R	*T. sebifera*	0	0.03	84.44	51.3	29.3	19.4	0.189	3.501	18.529
11175	R	*T. sebifera*	0	0.0256	85.92	61.5	18.9	19.6	0.144	1.762	12.241
11200	R	*T. sebifera*	0	0.0295	85.75	48.4	28.9	22.7	0.182	2.480	13.643
11225	R	*T. sebifera*	0	0.06	81.15	55.3	26.3	18.4	0.158	1.960	12.395
11250	R	*T. sebifera*	0	0.0716	75.79	23.1	40.6	36.3	0.197	2.841	14.446
11275	R	*T. sebifera*	0	0.1037	74.05	65.8	14.9	19.3	0.147	2.854	19.361
11300	R	*T. sebifera*	0	0.0814	57.24	33.5	34.3	32.2	0.320	4.026	12.584

Appendix B (I). Data from germination experiment in the growth chamber.

Species	Replicate	Salinity level (ppt)	Final % germination
B. halimifolia	1	0	70
B. halimifolia	1	10	78
B. halimifolia	1	20	76
B. halimifolia	1	30	10
B. halimifolia	2	0	92
B. halimifolia	2	10	86
B. halimifolia	2	20	56
B. halimifolia	2	30	8
B. halimifolia	3	0	92
B. halimifolia	3	10	88
B. halimifolia	3	20	76
B. halimifolia	3	30	14
M. cerifera	1	0	73.3
M. cerifera	1	10	73.33
M. cerifera	1	20	36.66
M. cerifera	1	30	0
M. cerifera	2	0	70
M. cerifera	2	10	90
M. cerifera	2	20	53.33
M. cerifera	2	30	0
M. cerifera	3	0	75
M. cerifera	3	10	85
M. cerifera	3	20	45
M. cerifera	3	30	3.33
T. sebifera	1	0	45.31
T. sebifera	1	10	4.68
T. sebifera	1	20	32.81
T. sebifera	1	30	9.37
T. sebifera	2	0	3.12
T. sebifera	2	10	0
T. sebifera	2	20	0
T. sebifera	2	30	0
T. sebifera	3	0	0
T. sebifera	3	10	0
T. sebifera	3	20	0
T. sebifera	3	30	0

Appendix B (II). Data from germination experiment in the greenhouse.

Species	Soil location	Salinity level (ppt)	Replicate	Final % germination
B. halimifolia	1	0	1	5.55
B. halimifolia	1	0	2	33.33
B. halimifolia	1	0	3	58.33
B. halimifolia	1	10	1	0.00
B. halimifolia	1	10	2	0.00
B. halimifolia	1	10	3	0.00
B. halimifolia	1	20	1	0.00
B. halimifolia	1	20	2	0.00
B. halimifolia	1	20	3	0.00
B. halimifolia	1	30	1	8.33
B. halimifolia	1	30	2	2.77
B. halimifolia	1	30	3	5.56
B. halimifolia	2	0	1	22.22
B. halimifolia	2	0	2	16.67
B. halimifolia	2	0	3	22.22
B. halimifolia	2	10	1	0.00
B. halimifolia	2	10	2	0.00
B. halimifolia	2	10	3	0.00
B. halimifolia	2	20	1	0.00
B. halimifolia	2	20	2	0.00
B. halimifolia	2	20	3	0.00
B. halimifolia	2	30	1	0.00
B. halimifolia	2	30	2	0.00
B. halimifolia	2	30	3	0.00
B. halimifolia	3	0	1	72.22
B. halimifolia	3	0	2	55.56
B. halimifolia	3	0	3	91.67
B. halimifolia	3	10	1	13.89
B. halimifolia	3	10	2	22.22
B. halimifolia	3	10	3	16.67
B. halimifolia	3	20	1	0.00
B. halimifolia	3	20	2	0.00
B. halimifolia	3	20	3	2.78
B. halimifolia	3	30	1	0.00
B. halimifolia	3	30	2	0.00
B. halimifolia	3	30	3	0.00
B. halimifolia	4	0	1	38.89
B. halimifolia	4	0	2	38.89
B. halimifolia	4	0	3	52.78
B. halimifolia	4	10	1	13.89
B. halimifolia	4	10	2	16.67
B. halimifolia	4	10	3	36.11
B. halimifolia	4	20	1	19.44
B. halimifolia	4	20	2	2.78
B. halimifolia	4	20	3	5.56
B. halimifolia	4	30	1	0.00
B. halimifolia	4	30	2	0.00
B. halimifolia	4	30	3	0.00
B. halimifolia	5	0	1	75.00
B. halimifolia	5	0	2	72.22

Appendix B (II), continued

B. halimifolia	5	0	3	44.44
B. halimifolia	5	10	1	27.78
B. halimifolia	5	10	2	19.44
B. halimifolia	5	10	3	11.11
B. halimifolia	5	20	1	13.89
B. halimifolia	5	20	2	2.78
B. halimifolia	5	20	3	8.33
B. halimifolia	5	30	1	0.00
B. halimifolia	5	30	2	11.11
B. halimifolia	5	30	3	0.00
M. cerifera	0	1	1	0.00
M. cerifera	0	1	2	0.00
M. cerifera	0	1	3	2.17
M. cerifera	10	1	1	0.00
M. cerifera	10	1	2	0.00
M. cerifera	10	1	3	0.00
M. cerifera	20	1	1	0.00
M. cerifera	20	1	2	0.00
M. cerifera	20	1	3	0.00
M. cerifera	30	1	1	0.00
M. cerifera	30	1	2	0.00
M. cerifera	30	1	3	0.00
M. cerifera	0	2	1	36.96
M. cerifera	0	2	2	26.09
M. cerifera	0	2	3	47.83
M. cerifera	10	2	1	8.70
M. cerifera	10	2	2	19.57
M. cerifera	10	2	3	17.39
M. cerifera	20	2	1	8.70
M. cerifera	20	2	2	0.00
M. cerifera	20	2	3	0.00
M. cerifera	30	2	1	0.00
M. cerifera	30	2	2	0.00
M. cerifera	30	2	3	0.00
M. cerifera	0	3	1	8.70
M. cerifera	0	3	2	10.87
M. cerifera	0	3	3	13.04
M. cerifera	10	3	1	0.00
M. cerifera	10	3	2	0.00
M. cerifera	10	3	3	6.52
M. cerifera	20	3	1	6.52
M. cerifera	20	3	2	6.52
M. cerifera	20	3	3	2.17
M. cerifera	30	3	1	2.17
M. cerifera	30	3	2	2.17
M. cerifera	30	3	3	6.52
M. cerifera	0	4	1	36.96
M. cerifera	0	4	2	32.61
M. cerifera	0	4	3	6.52
M. cerifera	10	4	1	21.74
M. cerifera	10	4	2	15.22

Appendix B (II), continued

M. cerifera	10	4	3	8.70
M. cerifera	20	4	1	8.70
M. cerifera	20	4	2	13.04
M. cerifera	20	4	3	15.22
M. cerifera	30	4	1	0.00
M. cerifera	30	4	2	0.00
M. cerifera	30	4	3	0.00
M. cerifera	0	5	1	39.13
M. cerifera	0	5	2	67.39
M. cerifera	0	5	3	45.65
M. cerifera	10	5	1	23.91
M. cerifera	10	5	2	21.74
M. cerifera	10	5	3	13.04
M. cerifera	20	5	1	0.00
M. cerifera	20	5	2	0.00
M. cerifera	20	5	3	0.00
M. cerifera	30	5	1	0.00
M. cerifera	30	5	2	0.00
M. cerifera	30	5	3	0.00
T. sebifera	1	0	1	2.5
T. sebifera	1	0	2	0
T. sebifera	1	0	3	2.5
T. sebifera	1	10	1	0
T. sebifera	1	10	2	0
T. sebifera	1	10	3	0
T. sebifera	1	20	1	0
T. sebifera	1	20	2	0
T. sebifera	1	20	3	0
T. sebifera	1	30	1	0
T. sebifera	1	30	2	0
T. sebifera	1	30	3	0
T. sebifera	2	0	1	35
T. sebifera	2	0	2	25
T. sebifera	2	0	3	17.5
T. sebifera	2	10	1	27.5
T. sebifera	2	10	2	17.5
T. sebifera	2	10	3	32.5
T. sebifera	2	20	1	2.5
T. sebifera	2	20	2	0
T. sebifera	2	20	3	0
T. sebifera	2	30	1	0
T. sebifera	2	30	2	0
T. sebifera	2	30	3	0
T. sebifera	3	0	1	65
T. sebifera	3	0	2	40
T. sebifera	3	0	3	40
T. sebifera	3	10	1	7.5
T. sebifera	3	10	2	0
T. sebifera	3	10	3	15
T. sebifera	3	20	1	0
T. sebifera	3	20	2	0

Appendix B (II), continued

T. sebifera	3	20	3	0
T. sebifera	3	30	1	0
T. sebifera	3	30	2	20
T. sebifera	3	30	3	0
T. sebifera	4	0	1	55
T. sebifera	4	0	2	52.5
T. sebifera	4	0	3	57.5
T. sebifera	4	10	1	32.5
T. sebifera	4	10	2	42.5
T. sebifera	4	10	3	32.5
T. sebifera	4	20	1	15
T. sebifera	4	20	2	7.5
T. sebifera	4	20	3	2.5
T. sebifera	4	30	1	0
T. sebifera	4	30	2	0
T. sebifera	4	30	3	0
T. sebifera	5	0	1	42.5
T. sebifera	5	0	2	67.5
T. sebifera	5	0	3	25
T. sebifera	5	10	1	15
T. sebifera	5	10	2	25
T. sebifera	5	10	3	20
T. sebifera	5	20	1	0
T. sebifera	5	20	2	12.5
T. sebifera	5	20	3	0
T. sebifera	5	30	1	2.5
T. sebifera	5	30	2	20
T. sebifera	5	30	3	32.5

Appendix C (I). Growth data from greenhouse experiment that simulated the effects of tropical storms (canopy openness and storm surge) at simulated forest stands of GBNERR. WPF = Wet pine forest, PFW = Pine flatwoods, Rep = Replications, RGR = Relative growth rates

Forest type	Species	Rep	% Canopy openness	Storm surge	RGR (1)	RGR (2)	RGR (3)	RGR (4)	RGR (5)	RGR (6)	RGR (7)	RGR (8)	RGR (9)
WPF	*B. halimifolia*	1	100%	No Surge	0.011	0.007	0.004	0.009	0.004	0.003	0.002	0.001	0.000
WPF	*B. halimifolia*	2	100%	No Surge	0.016	0.028	0.023	0.006	0.003	0.001	0.002	0.001	0.001
WPF	*B. halimifolia*	3	100%	Surge	0.040	0.020	0.009	0.005	0.002	0.000	0.004	0.001	0.001
WPF	*B. halimifolia*	4	100%	No Surge	0.018	0.021	0.005	0.001	0.001	0.001	0.005	0.002	0.001
WPF	*B. halimifolia*	5	100%	No Surge	0.052	0.019	0.005	0.000	0.002	0.001	0.010	0.005	0.002
WPF	*B. halimifolia*	6	100%	Surge	0.035	0.011	0.001	0.004	0.003	0.001	0.011	0.004	0.001
WPF	*B. halimifolia*	7	100%	Surge	0.033	0.013	0.007	0.006	.	0.003	0.001	0.001	0.002
WPF	*B. halimifolia*	8	100%	Surge	0.038	0.007	-0.056	.	.	.	.	.	.
WPF	*B. halimifolia*	9	100%	Surge	0.050	0.029	0.013	0.006	0.001	0.000	0.002	0.001	0.001
WPF	*B. halimifolia*	10	100%	No Surge	0.041	0.028	0.014	0.007	0.001	0.002	0.004	0.001	0.001
WPF	*B. halimifolia*	1	30%	Surge	0.032	0.015	0.008	0.002	0.007	-0.006	0.001	0.005	0.009
WPF	*B. halimifolia*	2	30%	No Surge	0.021	0.007	0.014	.	.	.	.	.	.
WPF	*B. halimifolia*	3	30%	Surge	0.018	0.007	0.004	0.006	0.001	0.000	0.000	0.001	0.001
WPF	*B. halimifolia*	4	30%	No Surge	0.028	0.012	0.013	0.004	0.001	0.000	0.003	0.002	0.001
WPF	*B. halimifolia*	5	30%	No Surge	0.032	0.016	0.006	.	.	.	.	.	.
WPF	*B. halimifolia*	6	30%	Surge	0.014	0.006	0.005	0.002	.	.	.	.	.
WPF	*B. halimifolia*	7	30%	Surge	0.025	0.011	0.005	0.003	0.000	0.000	0.006	0.009	0.008
WPF	*B. halimifolia*	8	30%	No Surge	0.021	0.006	0.000	.	.	.	.	.	.
WPF	*B. halimifolia*	9	30%	Surge	0.030	0.007	0.005	0.002	0.002	0.001	0.001	.	.
WPF	*B. halimifolia*	10	30%	Surge	0.019	0.005	0.003	0.000	0.000	0.000	0.002	0.006	0.006
WPF	*M. cerifera*	1	100%	No Surge	0.014	0.005	0.011	0.006	0.000	0.009	0.004	0.002	0.000
WPF	*M. cerifera*	2	100%	No Surge	0.018	0.012	0.011	0.002	0.000	0.004	0.005	0.004	0.004
WPF	*M. cerifera*	3	100%	Surge	0.030	0.013	-0.002	0.007	0.000	0.007	0.004	0.006	0.006
WPF	*M. cerifera*	4	100%	No Surge	0.024	0.018	0.006	0.000	0.002	0.002	0.002	0.000	0.002
WPF	*M. cerifera*	5	100%	No Surge	0.020	0.007	0.009	0.005	0.000	0.005	0.005	0.005	0.002
WPF	*M. cerifera*	6	100%	Surge	0.018	0.005	0.004	0.001	0.000	0.010	0.010	0.002	0.000
WPF	*M. cerifera*	7	100%	Surge	0.025	0.002	-0.055	.	.	.	.	.	.
WPF	*M. cerifera*	8	100%	Surge	0.019	0.004	-0.062	.	.	.	.	.	.
WPF	*M. cerifera*	9	100%	Surge	0.032	0.007	-0.001	0.000	0.001	0.003	0.002	0.001	0.000
WPF	*M. cerifera*	10	100%	No Surge	0.023	0.015	0.010	0.002	0.001	0.007	0.006	0.006	0.003

Appendix C (I). Growth data from greenhouse experiment, continued

WPF	*M. cerifera*	1	30%	Surge	0.013	0.000	.	.	.	.	.	.	.
WPF	*M. cerifera*	2	30%	No Surge	0.028	0.008	0.004	.	.	.	.	.	.
WPF	*M. cerifera*	3	30%	Surge	.	.	.	.	.	.	.	.	.
WPF	*M. cerifera*	4	30%	No Surge	0.019	0.006	0.002	0.003	0.001	0.002	0.006	0.008	0.007
WPF	*M. cerifera*	5	30%	No Surge	0.024	0.001	0.001	0.000	0.001	.	.	.	.
WPF	*M. cerifera*	6	30%	Surge	0.008	0.007	.	.	.	.	.	.	.
WPF	*M. cerifera*	7	30%	No Surge	0.008	0.003	0.004	0.000	0.000	0.003	0.003	.	.
WPF	*M. cerifera*	8	30%	No Surge	0.002	0.003	0.005	0.001	0.004	0.003	0.003	0.005	0.002
WPF	*M. cerifera*	9	30%	Surge	0.019	0.005	-0.004	.	.	.	.	.	.
WPF	*M. cerifera*	10	30%	Surge	0.015	0.005	0.001	0.001	0.001	0.002	0.007	0.006	0.002
WPF	*T. sebifera*	1	100%	No Surge	0.019	0.005	0.006	0.001	0.000	-0.005	0.006	0.001	0.002
WPF	*T. sebifera*	2	100%	No Surge	0.017	0.026	0.009	0.000	0.000	-0.002	0.004	0.003	0.001
WPF	*T. sebifera*	3	100%	Surge	0.026	0.018	0.002	0.000	0.000	-0.011	0.000	-0.001	0.000
WPF	*T. sebifera*	4	100%	No Surge	0.026	0.026	0.003	0.000	0.000	0.000	0.004	0.004	0.001
WPF	*T. sebifera*	5	100%	No Surge	0.021	0.024	0.008	0.002	0.000	0.000	0.000	0.002	0.001
WPF	*T. sebifera*	6	100%	Surge	0.031	0.016	-0.004	0.001	-0.001	0.000	-0.003	0.000	-0.008
WPF	*T. sebifera*	7	100%	Surge	0.021	0.012	-0.071	0.008	.	.	.	.	.
WPF	*T. sebifera*	8	100%	Surge	0.010	0.007	-0.062	0.003	.	.	.	.	.
WPF	*T. sebifera*	9	100%	Surge	0.027	0.022	-0.007	-0.009	-0.003	-0.005	0.004	0.003	0.001
WPF	*T. sebifera*	10	100%	No Surge	0.017	0.020	0.009	0.001	0.000	0.001	0.003	0.003	0.000
WPF	*T. sebifera*	1	30%	Surge	0.032	0.018	-0.051	.	.	.	.	.	.
WPF	*T. sebifera*	2	30%	No Surge	0.030	0.017	0.004	0.000	0.000	0.004	0.001	0.001	0.003
WPF	*T. sebifera*	3	30%	Surge	0.017	0.016	-0.102	0.000	.	.	.	.	.
WPF	*T. sebifera*	4	30%	No Surge	0.021	0.013	0.002	0.000	-0.001	0.000	0.000	0.000	0.001
WPF	*T. sebifera*	5	30%	No Surge	0.025	0.016	0.008	0.001	0.000	0.000	0.005	0.003	0.002
WPF	*T. sebifera*	6	30%	Surge	0.015	-0.009	0.012	.	.	.	.	.	.
WPF	*T. sebifera*	7	30%	No Surge	0.015	0.014	0.006	0.000	0.003	-0.005	0.005	.	.
WPF	*T. sebifera*	8	30%	No Surge	0.024	0.015	0.002	0.001	0.000	0.000	0.002	.	.
WPF	*T. sebifera*	9	30%	Surge	0.021	0.009	0.003	-0.001	.	.	.	.	.
WPF	*T. sebifera*	10	30%	Surge	0.025	0.014	0.001	0.000	0.000	0.000	0.004	0.001	0.003
MHF	*B. halimifolia*	1	70%	No Surge	0.010	0.004	0.005	0.002	0.001	0.005	0.008	0.010	0.002
MHF	*B. halimifolia*	2	70%	No Surge	0.002	0.000	0.005	0.004	0.003	0.002	0.001	0.003	0.003
MHF	*B. halimifolia*	3	70%	Surge	0.013	0.001	-0.003	0.003	0.001	0.009	0.016	0.004	0.003
MHF	*B. halimifolia*	4	70%	No Surge	0.028	0.009	0.012	0.003	0.001	-0.001	0.014	0.014	0.007
MHF	*B. halimifolia*	5	70%	Surge	0.011	0.005	0.003	0.000	0.000	0.006	0.019	0.010	0.001
MHF	*B. halimifolia*	6	70%	Surge	0.021	0.010	0.002	0.002	0.000	0.001	0.003	0.005	0.009

Appendix C (I). Growth data from greenhouse experiment, continued

MHF	*B. halimifolia*	7	70%	No Surge	0.013	0.005	0.004	0.003	0.002	0.002	0.004	0.002	0.001
MHF	*B. halimifolia*	8	70%	Surge	0.018	0.003	0.001	-0.038	.	.	.	.	.
MHF	*B. halimifolia*	9	70%	Surge	0.032	0.006	0.005	0.008	0.003	0.008	0.016	0.010	0.004
MHF	*B. halimifolia*	10	70%	No Surge	0.011	0.008	0.007	0.004	0.001	0.001	0.008	0.009	0.001
MHF	*B. halimifolia*	1	10%	Surge	.	.	.	.	.	.	.	.	.
MHF	*B. halimifolia*	2	10%	No Surge	0.006	0.004	.	.	.	.	.	.	.
MHF	*B. halimifolia*	3	10%	Surge	0.022	0.009	.	.	.	.	.	.	.
MHF	*B. halimifolia*	4	10%	Surge	-0.009	0.028	0.011	0.002	.	.	.	.	.
MHF	*B. halimifolia*	5	10%	No Surge	0.004	0.000	0.001	0.002	0.002	0.000	0.001	.	.
MHF	*B. halimifolia*	6	10%	No Surge	.	.	.	.	.	.	.	.	.
MHF	*B. halimifolia*	7	10%	Surge	0.008	0.008	.	.	.	.	.	.	.
MHF	*B. halimifolia*	8	10%	Surge	.	.	.	.	.	.	.	.	.
MHF	*B. halimifolia*	9	10%	No Surge	.	.	.	.	.	.	.	.	.
MHF	*B. halimifolia*	10	10%	No Surge	.	.	.	.	.	.	.	.	.
MHF	*M. cerifera*	1	70%	No Surge	0.018	0.007	0.002	0.000	0.000	0.000	0.001	0.004	0.001
MHF	*M. cerifera*	2	70%	No Surge	0.038	0.007	0.004	0.000	0.001	0.006	0.007	0.010	0.002
MHF	*M. cerifera*	3	70%	Surge	0.025	0.009	.	.	.	.	.	.	.
MHF	*M. cerifera*	4	70%	No Surge	0.043	0.007	0.004	0.001	0.000	0.001	0.002	0.003	0.003
MHF	*M. cerifera*	5	70%	Surge	0.037	0.008	0.002	0.000	0.000	0.002	0.005	0.006	0.001
MHF	*M. cerifera*	6	70%	Surge	0.032	0.008	0.002	-0.004	.	.	.	.	.
MHF	*M. cerifera*	7	70%	No Surge	0.037	0.002	0.006	0.000	0.001	0.003	0.002	0.005	0.001
MHF	*M. cerifera*	8	70%	Surge	0.037	0.017	.	.	.	.	.	.	.
MHF	*M. cerifera*	9	70%	Surge	0.051	0.010	0.000	0.001	0.000	0.004	0.006	0.013	0.004
MHF	*M. cerifera*	10	70%	No Surge	0.032	0.007	0.003	0.002	0.000	0.004	0.008	0.009	0.005
MHF	*M. cerifera*	1	10%	Surge	0.004	.	.	.	.	.	.	.	.
MHF	*M. cerifera*	2	10%	No Surge	0.002	0.003	-0.026	.	.	.	.	.	.
MHF	*M. cerifera*	3	10%	Surge	0.015	0.005	.	.	.	.	.	.	.
MHF	*M. cerifera*	4	10%	No Surge	0.037	.	.	.	.	.	.	.	.
MHF	*M. cerifera*	5	10%	Surge	.	.	.	.	.	.	.	.	.
MHF	*M. cerifera*	6	10%	No Surge	0.007	.	.	.	.	.	.	.	.
MHF	*M. cerifera*	7	10%	Surge	0.007	-0.001	.	.	.	.	.	.	.
MHF	*M. cerifera*	8	10%	Surge	0.003	.	.	.	.	.	.	.	.
MHF	*M. cerifera*	9	10%	No Surge	0.007	-0.022	.	.	.	.	.	.	.
MHF	*M. cerifera*	10	10%	No Surge	0.006	-0.008	0.000	0.000	0.005	.	.	.	.
MHF	*T. sebifera*	1	70%	No Surge	0.013	0.006	0.004	0.000	0.000	0.000	0.001	0.011	0.001
MHF	*T. sebifera*	2	70%	No Surge	0.012	0.001	0.000	0.000	0.001	0.000	0.001	0.001	-0.013

Appendix C (I). Growth data from greenhouse experiment, continued

MHF	*T. sebifera*	3	70%	Surge	0.014	0.003	0.002	0.000	0.000	0.001	0.003	0.007	0.001
MHF	*T. sebifera*	4	70%	No Surge	0.021	0.011	0.008	0.002	0.000	0.000	0.003	0.012	0.000
MHF	*T. sebifera*	5	70%	Surge	0.015	0.007	0.001	0.000	0.000	0.000	0.003	0.008	0.000
MHF	*T. sebifera*	6	70%	Surge	0.023	0.006	-0.018	-0.034	0.000	.	.	.	.
MHF	*T. sebifera*	7	70%	No Surge	0.013	0.006	0.003	0.000	0.001	0.001	0.006	0.007	0.001
MHF	*T. sebifera*	8	70%	Surge	0.007	0.004	-0.030	-0.029	.	.	.	.	.
MHF	*T. sebifera*	9	70%	Surge	0.016	-0.002	0.000	-0.051	.	.	.	.	.
MHF	*T. sebifera*	10	70%	No Surge	0.011	0.006	0.003	0.000	0.000	0.000	0.002	0.011	0.001
MHF	*T. sebifera*	1	10%	Surge	0.021	0.010	.	.	.	.	.	.	.
MHF	*T. sebifera*	2	10%	No Surge	0.018	0.002	.	.	.	.	.	.	.
MHF	*T. sebifera*	3	10%	Surge	0.009	0.002	.	.	.	.	.	.	.
MHF	*T. sebifera*	4	10%	No Surge	0.017	-0.003	0.002	0.000	0.000	-0.001	-0.026	.	.
MHF	*T. sebifera*	5	10%	Surge	0.016	0.005	.	.	.	.	.	.	.
MHF	*T. sebifera*	6	10%	No Surge	0.019	0.003	0.005	0.000	0.000	0.001	0.002	0.000	0.001
MHF	*T. sebifera*	7	10%	Surge	0.010	0.008	0.000	.	.	.	.	.	.
MHF	*T. sebifera*	8	10%	Surge	0.024	0.011	.	.	.	.	.	.	.
MHF	*T. sebifera*	9	10%	No Surge	0.023	0.011	0.002	0.001	0.000	0.001	0.002	0.004	0.000
MHF	*T. sebifera*	10	10%	No Surge	0.022	0.009	0.003	0.002	0.001	0.000	0.002	0.002	0.001

Appendix C (II). Biomass data from greenhouse experiment that simulated the effects of tropical storms (canopy openness and storm surge) at simulated forest stands of GBNERR. WPF = Wet pine forest, PFW = Pine flatwoods

Forest type	canopy openness	Storm surge	Replicate	Species	Dry shoot biomass (gm)	Dry root biomass (gm)	Total (gm)
WPF	100%	No Surge	1	*B. halimifolia*	64.061	15.574	79.635
WPF	100%	No Surge	2	*B. halimifolia*	15.592	4.759	20.351
WPF	100%	No Surge	3	*B. halimifolia*	52.4	13.065	65.465
WPF	100%	No Surge	4	*B. halimifolia*	1.811	0.598	2.409
WPF	100%	No Surge	5	*B. halimifolia*	24.6	8.2	32.8
WPF	100%	Surge	1	*B. halimifolia*	48.9	39.749	88.649
WPF	100%	Surge	2	*B. halimifolia*	28	5.4	33.4
WPF	100%	Surge	3	*B. halimifolia*	14.7	1.994	16.694
WPF	100%	Surge	4	*B. halimifolia*	.	.	.
WPF	100%	Surge	5	*B. halimifolia*	94.3	39.1	133.4
WPF	100%	No Surge	1	*M. cerifera*	6.989	2.162	9.151
WPF	100%	No Surge	2	*M. cerifera*	7.234	2.321	9.555
WPF	100%	No Surge	3	*M. cerifera*	10.794	5.1	15.894
WPF	100%	No Surge	4	*M. cerifera*	20.984	9.647	30.631
WPF	100%	No Surge	5	*M. cerifera*	6.034	3.1	9.134
WPF	100%	Surge	1	*M. cerifera*	16.738	6.5	23.238
WPF	100%	Surge	2	*M. cerifera*	5.168	3.5	8.668
WPF	100%	Surge	3	*M. cerifera*	.	.	.
WPF	100%	Surge	4	*M. cerifera*	.	.	.
WPF	100%	Surge	5	*M. cerifera*	0.313	0.055	0.368
WPF	100%	No Surge	1	*T. sebifera*	0.777	0.9	1.677
WPF	100%	No Surge	2	*T. sebifera*	17.2	13.863	31.063
WPF	100%	No Surge	3	*T. sebifera*	1.946	1.929	3.875
WPF	100%	No Surge	4	*T. sebifera*	16.048	12.844	28.892
WPF	100%	No Surge	5	*T. sebifera*	8.6	7.7	16.3
WPF	100%	Surge	1	*T. sebifera*	4.39	4.081	8.471
WPF	100%	Surge	2	*T. sebifera*	8.331	4.9	13.231
WPF	100%	Surge	3	*T. sebifera*	.	.	.
WPF	100%	Surge	4	*T. sebifera*	.	.	.
WPF	100%	Surge	5	*T. sebifera*	3.048	1.621	4.669
WPF	70%	No Surge	1	*B. halimifolia*	.	.	.
WPF	70%	No Surge	2	*B. halimifolia*	2.383	0.109	2.492

Appendix C (II). Biomass data from greenhouse experiment, continued

WPF	70%	No Surge	3	*B. halimifolia*	.	.	.
WPF	70%	no Surge	4	*B. halimifolia*	0.492	0.112	0.604
WPF	70%	No Surge	5	*B. halimifolia*	.	.	.
WPF	70%	Surge	1	*B. halimifolia*	0.805	0.053	0.858
WPF	70%	Surge	2	*B. halimifolia*	16.6	2.422	19.022
WPF	70%	Surge	3	*B. halimifolia*	.	.	.
WPF	70%	Surge	4	*B. halimifolia*	.	.	.
WPF	70%	Surge	5	*B. halimifolia*	0.585	0.054	0.639
WPF	70%	No Surge	1	*M. cerifera*	.	.	.
WPF	70%	No Surge	2	*M. cerifera*	0.299	0.006	0.305
WPF	70%	No Surge	3	*M. cerifera*	.	.	.
WPF	70%	No Surge	4	*M. cerifera*	.	.	.
WPF	70%	No Surge	5	*M. cerifera*	0.327	0.042	0.369
WPF	70%	Surge	1	*M. cerifera*	.	.	.
WPF	70%	Surge	2	*M. cerifera*	.	.	.
WPF	70%	Surge	3	*M. cerifera*	.	.	.
WPF	70%	Surge	4	*M. cerifera*	.	.	.
WPF	70%	Surge	5	*M. cerifera*	0.858	0.234	1.092
WPF	70%	No Surge	1	*T. sebifera*	5.1	1.2	6.3
WPF	70%	No Surge	2	*T. sebifera*	1.299	0.388	1.687
WPF	70%	No Surge	3	*T. sebifera*	15.4	4.955	20.355
WPF	70%	No Surge	4	*T. sebifera*	.	.	.
WPF	70%	No Surge	5	*T. sebifera*	.	.	.
WPF	70%	Surge	1	*T. sebifera*	.	.	.
WPF	70%	Surge	2	*T. sebifera*	.	.	.
WPF	70%	Surge	3	*T. sebifera*	.	.	.
WPF	70%	Surge	4	*T. sebifera*	.	.	.
WPF	70%	Surge	5	*T. sebifera*	10.5	3.096	13.596
MHF	30%	No Surge	1	*B. halimifolia*	15.2	4.142	19.342
MHF	30%	No Surge	2	*B. halimifolia*	12.1	3.7	15.8
MHF	30%	No Surge	3	*B. halimifolia*	1.975	0.569	2.544
MHF	30%	No Surge	4	*B. halimifolia*	5.5	3.312	8.812
MHF	30%	No Surge	5	*B. halimifolia*	10.7	5.781	16.481
MHF	30%	Surge	1	*B. halimifolia*	1.402	0.692	2.094
MHF	30%	Surge	2	*B. halimifolia*	10.2	4	14.2
MHF	30%	Surge	3	*B. halimifolia*	0.544	0.192	0.736

Appendix C (II). Biomass data from greenhouse experiment

MHF	30%	Surge	4	*B. halimifolia*	.	.	.
MHF	30%	Surge	5	*B. halimifolia*	13	4.734	17.734
MHF	30%	No Surge	1	*M. cerifera*	0.112	0.122	0.234
MHF	30%	No Surge	2	*M. cerifera*	2.728	1.3	4.028
MHF	30%	No Surge	3	*M. cerifera*	0.374	0.339	0.713
MHF	30%	No Surge	4	*M. cerifera*	0.632	0.767	1.399
MHF	30%	No Surge	5	*M. cerifera*	4.308	2.3	6.608
MHF	30%	Surge	1	*M. cerifera*	.	.	.
MHF	30%	Surge	2	*M. cerifera*	4.7	3.1	7.8
MHF	30%	Surge	3	*M. cerifera*	.	.	.
MHF	30%	Surge	4	*M. cerifera*	.	.	.
MHF	30%	Surge	5	*M. cerifera*	4.4	1.908	6.308
MHF	30%	No Surge	1	*T. sebifera*	7.228	7.2	14.428
MHF	30%	No Surge	2	*T. sebifera*	1.155	0.191	1.346
MHF	30%	No Surge	3	*T. sebifera*	10.5	10.811	21.311
MHF	30%	No Surge	4	*T. sebifera*	5.9	6.9	12.8
MHF	30%	No Surge	5	*T. sebifera*	5.5	6.5	12
MHF	30%	Surge	1	*T. sebifera*	5.531	4.766	10.297
MHF	30%	Surge	2	*T. sebifera*	3.3	3.321	6.621
MHF	30%	Surge	3	*T. sebifera*	.	.	.
MHF	30%	Surge	4	*T. sebifera*	.	.	.
MHF	30%	Surge	5	*T. sebifera*	.	.	.
MHF	90%	No Surge	1	*B. halimifolia*	.	.	.
MHF	90%	No Surge	2	*B. halimifolia*	.	.	.
MHF	90%	No Surge	3	*B. halimifolia*	.	.	.
MHF	90%	No Surge	4	*B. halimifolia*	.	.	.
MHF	90%	No Surge	5	*B. halimifolia*	.	.	.
MHF	90%	Surge	1	*B. halimifolia*	.	.	.
MHF	90%	Surge	2	*B. halimifolia*	.	.	.
MHF	90%	Surge	3	*B. halimifolia*	.	.	.
MHF	90%	Surge	4	*B. halimifolia*	.	.	.
MHF	90%	Surge	5	*B. halimifolia*	.	.	.
MHF	90%	No Surge	1	*M. cerifera*	.	.	.
MHF	90%	No Surge	2	*M. cerifera*	.	.	.
MHF	90%	No Surge	3	*M. cerifera*	.	.	.
MHF	90%	No Surge	4	*M. cerifera*	.	.	.

Appendix C (II). Biomass data from greenhouse experiment

MHF	90%	No Surge	5	*M. cerifera*	.	.	.
MHF	90%	Surge	1	*M. cerifera*	.	.	.
MHF	90%	Surge	2	*M. cerifera*	.	.	.
MHF	90%	Surge	3	*M. cerifera*	.	.	.
MHF	90%	Surge	4	*M. cerifera*	.	.	.
MHF	90%	Surge	5	*M. cerifera*	.	.	.
MHF	90%	No Surge	1	*T. sebifera*	.	.	.
MHF	90%	No Surge	2	*T. sebifera*	.	.	.
MHF	90%	No Surge	3	*T. sebifera*	1.151	0.274	1.425
MHF	90%	No Surge	4	*T. sebifera*	1.184	0.259	1.443
MHF	90%	No Surge	5	*T. sebifera*	1.51	0.314	1.824
MHF	90%	Surge	1	*T. sebifera*	.	.	.
MHF	90%	Surge	2	*T. sebifera*	.	.	.
MHF	90%	Surge	3	*T. sebifera*	.	.	.
MHF	90%	Surge	4	*T. sebifera*	.	.	.
MHF	90%	Surge	5	*T. sebifera*	.	.	.

Appendix D (I). Raw data from greenhouse experiment (Chapter 5). WPF = Wet pine forest, PFW = PFW, (+) = without fungicide, (–) = with fungicide, Rep = replication, % Colon = colonization

Soil source	Species	VAM	Rep	Sample	Negative score VAM	hyphae	arbuscule	vesicle	Number of intersections	% Colon	Total biomass (gm)
WPF	*B. halimifolia*	(+)	1	a	68	10	2	0	80	17.5	0.188
WPF	*B. halimifolia*	(+)	1	b	38	17	11	14	80	83.75	.
WPF	*B. halimifolia*	(+)	1	c	70	3	2	5	80	21.25	.
WPF	*B. halimifolia*	(+)	1	d	76	2	1	1	80	7.5	.
WPF	*B. halimifolia*	(+)	1	e	68	7	3	2	80	21.25	.
WPF	*B. halimifolia*	(+)	1	f	78	1	1	0	80	3.75	.
WPF	*B. halimifolia*	(+)	1	g	69	6	3	2	80	20	.
WPF	*B. halimifolia*	(+)	1	h	65	7	6	2	80	28.75	.
WPF	*B. halimifolia*	(+)	1	i	80	0	0	0	80	0	.
WPF	*B. halimifolia*	(+)	2	a	72	5	2	1	80	13.75	1.26
WPF	*B. halimifolia*	(+)	2	b	67	4	4	5	80	27.5	.
WPF	*B. halimifolia*	(+)	2	c	73	4	3	0	80	12.5	.
WPF	*B. halimifolia*	(+)	2	d	50	12	10	8	80	60	.
WPF	*B. halimifolia*	(+)	2	e	63	8	7	2	80	32.5	.
WPF	*B. halimifolia*	(+)	2	f	73	5	2	0	80	11.25	.
WPF	*B. halimifolia*	(+)	2	g	51	9	20	0	80	61.25	.
WPF	*B. halimifolia*	(+)	2	h	75	1	4	0	80	11.25	.
WPF	*B. halimifolia*	(+)	2	i	67	11	2	0	80	18.75	.
WPF	*B. halimifolia*	(+)	3	a	72	6	2	0	80	12.5	0.115
WPF	*B. halimifolia*	(+)	3	b	57	18	2	3	80	35	.
WPF	*B. halimifolia*	(+)	3	c	65	13	2	0	80	21.25	.
WPF	*B. halimifolia*	(+)	3	d	69	8	3	0	80	17.5	.
WPF	*B. halimifolia*	(+)	3	e	71	7	2	0	80	13.75	.
WPF	*B. halimifolia*	(+)	3	f	67	6	4	3	80	25	.
WPF	*B. halimifolia*	(+)	4	a	78	2	0	0	80	2.5	.
WPF	*B. halimifolia*	(+)	4	b	31	17	16	16	80	101.25	0.043
WPF	*B. halimifolia*	(+)	4	c	80	0	0	0	80	0	.
WPF	*B. halimifolia*	(+)	4	d	75	3	2	0	80	8.75	.
WPF	*B. halimifolia*	(+)	4	e	57	9	8	6	80	46.25	.
WPF	*B. halimifolia*	(+)	4	f	67	7	6	0	80	23.75	.
WPF	*B. halimifolia*	(-)	1	a	78	2	0	0	80	2.5	0.159
WPF	*B. halimifolia*	(-)	1	b	79	1	0	0	80	1.25	.

Appendix D (I). Raw data from greenhouse experiment (Chapter 5), continued

WPF	*B. halimifolia*	(-)	1	c	78	2	0	0	80	2.5	.	
WPF	*B. halimifolia*	(-)	1	d	74	3	2	1	80	11.25	.	
WPF	*B. halimifolia*	(-)	1	e	80	0	0	0	80	0	.	
WPF	*B. halimifolia*	(-)	1	f	80	0	0	0	80	0	.	
WPF	*B. halimifolia*	(-)	2	a	75	3	2	0	80	8.75	0.41	
WPF	*B. halimifolia*	(-)	2	b	80	0	0	0	80	0	.	
WPF	*B. halimifolia*	(-)	2	c	75	3	1	1	80	8.75	.	
WPF	*B. halimifolia*	(-)	2	d	74	5	1	0	80	8.75	.	
WPF	*B. halimifolia*	(-)	2	e	76	4	0	0	80	5	.	
WPF	*B. halimifolia*	(-)	2	f	71	5	4	0	80	16.25	.	
WPF	*B. halimifolia*	(-)	2	g	80	0	0	0	80	0	.	
WPF	*B. halimifolia*	(-)	3	a	80	0	0	0	80	0	0.048	
WPF	*B. halimifolia*	(-)	3	b	80	0	0	0	80	0	.	
WPF	*B. halimifolia*	(-)	3	c	77	3	0	0	80	3.75	.	
WPF	*B. halimifolia*	(-)	3	d	79	1			80	1.25	.	
WPF	*B. halimifolia*	(-)	3	e	80	0	0	0	80	0	.	
WPF	*B. halimifolia*	(-)	3	f	80	0	0	0	80	0	.	
WPF	*B. halimifolia*	(-)	4	.	.	.	.		.	.	.	
WPF	*M. cerifera*	(+)	1	a	78	2	0	0	80	2.5	0.265	
WPF	*M. cerifera*	(+)	1	b	80	0	0	0	80	0	.	
WPF	*M. cerifera*	(+)	1	c	65	12	3	0	80	22.5	.	
WPF	*M. cerifera*	(+)	1	d	77	2	1	0	80	5	.	
WPF	*M. cerifera*	(+)	1	e	79	1	0	0	80	1.25	.	
WPF	*M. cerifera*	(+)	2	a	80	0	0	0	80	0	0.171	
WPF	*M. cerifera*	(+)	2	b	80	0	0	0	80	0	.	
WPF	*M. cerifera*	(+)	2	c	80	0	0	0	80	0	.	
WPF	*M. cerifera*	(+)	2	d	76	4	0	0	80	5	.	
WPF	*M. cerifera*	(+)	3	a	78	2	0	0	80	2.5	0.747	
WPF	*M. cerifera*	(+)	3	b	80	0	0	0	80	0	.	
WPF	*M. cerifera*	(+)	3	c	80	0	0	0	80	0	.	
WPF	*M. cerifera*	(+)	3	d	80	0	0	0	80	0	.	
WPF	*M. cerifera*	(+)	3	e	80	0	0	0	80	0	.	
WPF	*M. cerifera*	(+)	3	f	80	0	0	0	80	0	.	
WPF	*M. cerifera*	(+)	3	g	56	10	6	8	80	47.5	.	
WPF	*M. cerifera*	(+)	4	a	76	4	0	0	80	5	1.006	
WPF	*M. cerifera*	(+)	4	b	54	10	6	10	80	52.5	.	

Appendix D (I). Raw data from greenhouse experiment (Chapter 5), continued

WPF	*M. cerifera*	(+)	4	c	74	6	0	0	80	7.5	.
WPF	*M. cerifera*	(+)	4	d	68	8	2	2	80	20	.
WPF	*M. cerifera*	(-)	1	a	80	0	0	0	80	0	0.749
WPF	*M. cerifera*	(-)	1	b	76	3	1	0	80	6.25	.
WPF	*M. cerifera*	(-)	1	c	72	7	1	0	80	11.25	.
WPF	*M. cerifera*	(-)	1	d	78	2	0	0	80	2.5	.
WPF	*M. cerifera*	(-)	1	e	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	1	f	79	1	0	0	80	1.25	.
WPF	*M. cerifera*	(-)	1	g	79	1	0	0	80	1.25	.
WPF	*M. cerifera*	(-)	1	h	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	1	i	76	3	1	0	80	6.25	.
WPF	*M. cerifera*	(-)	2	a	80	0	0	0	80	0	0.202
WPF	*M. cerifera*	(-)	2	b	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	2	c	79	1	0	0	80	1.25	.
WPF	*M. cerifera*	(-)	2	d	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	2	e	78	2	0	0	80	2.5	.
WPF	*M. cerifera*	(-)	2	f	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	2	g	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	2	h	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	2	i	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	3	a	80	0	0	0	80	0	1.487
WPF	*M. cerifera*	(-)	3	b	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	3	c	79	1	0	0	80	1.25	.
WPF	*M. cerifera*	(-)	3	d	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	3	e	77	2	0	1	80	5	.
WPF	*M. cerifera*	(-)	3	f	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	3	g	79	1	0	0	80	1.25	.
WPF	*M. cerifera*	(-)	3	h	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	3	i	80	0	0	0	80	0	.
WPF	*M. cerifera*	(-)	4	.	.	.	.	.	.	.	.
WPF	*T. sebifera*	(+)	1	a	67	10	3	0	80	20	0.535
WPF	*T. sebifera*	(+)	1	b	58	13	6	3	80	38.75	.
WPF	*T. sebifera*	(+)	1	c	60	12	6	2	80	35	.
WPF	*T. sebifera*	(+)	1	d	49	15	14	2	80	58.75	.
WPF	*T. sebifera*	(+)	1	e	56	12	8	4	80	45	.
WPF	*T. sebifera*	(+)	2	a	59	5	6	10	80	46.25	0.717

Appendix D (I). Raw data from greenhouse experiment (Chapter 5), continued

WPF	*T. sebifera*	(+)	2	b	80	0	0	0	80	0	.
WPF	*T. sebifera*	(+)	2	c	67	2	6	5	80	30	.
WPF	*T. sebifera*	(+)	2	d	57	10	8	5	80	45	.
WPF	*T. sebifera*	(+)	2	e	58	8	7	7	80	45	.
WPF	*T. sebifera*	(+)	2	f	78	2	0	0	80	2.5	.
WPF	*T. sebifera*	(+)	2	g	79	1	0	0	80	1.25	.
WPF	*T. sebifera*	(+)	3	a	46	18	11	5	80	62.5	0.374
WPF	*T. sebifera*	(+)	3	b	41	13	12	14	80	81.25	.
WPF	*T. sebifera*	(+)	3	c	46	13	10	11	80	68.75	.
WPF	*T. sebifera*	(+)	3	d	65	5	4	6	80	31.25	.
WPF	*T. sebifera*	(+)	3	e	59	9	5	7	80	41.25	.
WPF	*T. sebifera*	(+)	3	f	63	11	3	3	80	28.75	.
WPF	*T. sebifera*	(+)	3	g	79	1	0	0	80	1.25	.
WPF	*T. sebifera*	(+)	4	a	25	24	16	15	80	107.5	0.302
WPF	*T. sebifera*	(+)	4	b	48	15	10	7	80	61.25	.
WPF	*T. sebifera*	(+)	4	c	48	15	10	7	80	61.25	.
WPF	*T. sebifera*	(+)	4	d	72	5	3	0	80	13.75	.
WPF	*T. sebifera*	(+)	4	e	58	9	8	5	80	43.75	.
WPF	*T. sebifera*	(+)	4	f	72	5	3	0	80	13.75	.
WPF	*T. sebifera*	(+)	4	g	77	3	0	0	80	3.75	.
WPF	*T. sebifera*	(-)	1	a	74	6	0	0	80	7.5	0.478
WPF	*T. sebifera*	(-)	1	b	74	5	1	0	80	8.75	.
WPF	*T. sebifera*	(-)	1	c	65	9	4	2	80	26.25	.
WPF	*T. sebifera*	(-)	1	d	71	6	1	2	80	15	.
WPF	*T. sebifera*	(-)	1	e	76	4	0	0	80	5	.
WPF	*T. sebifera*	(-)	1	f	62	7	5	6	80	36.25	.
WPF	*T. sebifera*	(-)	1	g	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	1	f	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	1	h	52	12	10	6	80	55	.
WPF	*T. sebifera*	(-)	2	a	80	0	0	0	80	0	0.311
WPF	*T. sebifera*	(-)	2	b	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	2	c	78	2	0	0	80	2.5	.
WPF	*T. sebifera*	(-)	2	d	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	2	e	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	2	f	77	3	0	0	80	3.75	.
WPF	*T. sebifera*	(-)	2	g	80	0	0	0	80	0	.

Appendix D (I). Raw data from greenhouse experiment (Chapter 5), continued

WPF	*T. sebifera*	(-)	2	f	75	5	0	0	80	6.25	.
WPF	*T. sebifera*	(-)	3	a	69	11	0	0	80	13.75	0.074
WPF	*T. sebifera*	(-)	3	b	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	3	c	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	3	d	60	17	3	0	80	28.75	.
WPF	*T. sebifera*	(-)	3	e	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	3	f	77	3	0	0	80	3.75	.
WPF	*T. sebifera*	(-)	4	a	80	0	0	0	80	0	0.144
WPF	*T. sebifera*	(-)	4	b	78	2	0	0	80	2.5	.
WPF	*T. sebifera*	(-)	4	c	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	4	d	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	4	e	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	4	f	74	3	0	3	80	11.25	.
WPF	*T. sebifera*	(-)	4	g	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	4	f	80	0	0	0	80	0	.
WPF	*T. sebifera*	(-)	4	g	80	0	0	0	80	0	.
PFW	*M. cerifera*	(+)	1	a	79	1	0	0	80	1.25	0.695
PFW	*M. cerifera*	(+)	1	b	68	7	3	2	80	21.25	.
PFW	*M. cerifera*	(+)	1	c	78	1	0	1	80	3.75	.
PFW	*M. cerifera*	(+)	1	d	77	2	0	1	80	5	.
PFW	*M. cerifera*	(+)	1	e	75	2	1	2	80	10	.
PFW	*M. cerifera*	(+)	2	a	80	0	0	0	80	0	1.46
PFW	*M. cerifera*	(+)	2	b	67	7	4	2	80	23.75	.
PFW	*M. cerifera*	(+)	2	c	80	0	0	0	80	0	.
PFW	*M. cerifera*	(+)	2	d	72	5	3	0	80	13.75	.
PFW	*M. cerifera*	(+)	2	e	79	0	0	1	80	2.5	.
PFW	*M. cerifera*	(+)	2	f	76	3	1	0	80	6.25	.
PFW	*M. cerifera*	(+)	2	g	80	0	0	0	80	0	.
PFW	*M. cerifera*	(+)	2	h	73	5	0	2	80	11.25	.
PFW	*M. cerifera*	(+)	3	a	76	4	0	0	80	5	0.322
PFW	*M. cerifera*	(+)	3	b	80	0	0	0	80	0	.
PFW	*M. cerifera*	(+)	3	c	69	8	3		80	17.5	.
PFW	*M. cerifera*	(+)	3	d	80	0	0	0	80	0	.
PFW	*M. cerifera*	(+)	3	e	80	0	0	0	80	0	.
PFW	*M. cerifera*	(+)	3	f	20	25	19	16	80	118.75	.
PFW	*M. cerifera*	(+)	3	g	80	0	0	0	80	0	.

Appendix D (I). Raw data from greenhouse experiment (Chapter 5), continued

PFW	*M. cerifera*	(+)	4	a	80	0	0	0	80	0	0.478
PFW	*M. cerifera*	(+)	4	b	80	0	0	0	80	0	.
PFW	*M. cerifera*	(+)	4	c	80	0	0	0	80	0	.
PFW	*M. cerifera*	(+)	4	d	71	4	2	3	80	17.5	.
PFW	*M. cerifera*	(+)	4	e	71	5	3	1	80	16.25	.
PFW	*M. cerifera*	(+)	4	f	80	0	0	0	80	0	.
PFW	*M. cerifera*	(+)	4	g	68	8	3	1	80	20	.
PFW	*M. cerifera*	(+)	4	h	80	0	0	0	80	0	.
PFW	*M. cerifera*	(+)	4	i	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	1	a	80	0	0	0	80	0	0.541
PFW	*M. cerifera*	(-)	1	b	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	1	c	79	1	0	0	80	1.25	.
PFW	*M. cerifera*	(-)	1	d	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	1	e	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	1	f	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	1	g	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	1	h	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	1	i	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	2	a	69	5	2	4	80	21.25	0.333
PFW	*M. cerifera*	(-)	2	b	76	2	1	1	80	7.5	.
PFW	*M. cerifera*	(-)	2	c	79	0	0	1	80	2.5	.
PFW	*M. cerifera*	(-)	2	d	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	2	e	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	2	f	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	2	g	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	2	h	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	2	i	77	3	0	0	80	3.75	.
PFW	*M. cerifera*	(-)	3	a	80	0	0	0	80	0	0.48
PFW	*M. cerifera*	(-)	3	b	80	0	0	0	80	0	.
PFW	*M. cerifera*	(-)	3	c	67	3	2	8	81	29.62	.
PFW	*M. cerifera*	(-)	3	d	80	0	0	0	82	2.43	.
PFW	*M. cerifera*	(-)	3	e	68	3	2	7	83	28.91	.
PFW	*M. cerifera*	(-)	3	f	68	2	1	9	84	30.95	.
PFW	*M. cerifera*	(-)	3	g	80	0	0	0	85	5.88	.
PFW	*M. cerifera*	(-)	3	h	67	4	2	7	86	32.55	.
PFW	*M. cerifera*	(-)	3	i	80	0	0	0	87	8.04	.

Appendix D (I). Raw data from greenhouse experiment (Chapter 5), continued

PFW	*M. cerifera*	(-)	4	a	71	5	0	4	88	23.86	0.869
PFW	*M. cerifera*	(-)	4	b	80	0	0	0	89	10.11	.
PFW	*M. cerifera*	(-)	4	c	76	3	0	1	80	6.25	.
PFW	*M. cerifera*	(-)	4	d	64	7	0	9	80	31.25	.
PFW	*M. cerifera*	(-)	4	e	75	3	0	2	80	8.75	.
PFW	*M. cerifera*	(-)	4	f	76	0	0	0	80	10	.
PFW	*M. cerifera*	(-)	4	g	75	3	0	4	80	8.75	.
PFW	*M. cerifera*	(-)	4	h	78	1	0	2	80	3.75	.
PFW	*M. cerifera*	(-)	4	i	77	0	0	1	80	7.5	.
PFW	*B. halimifolia*	(+)	1	a	66	7	4	3	80	26.25	1.55
PFW	*B. halimifolia*	(+)	1	b	71	5	4	0	80	16.25	.
PFW	*B. halimifolia*	(+)	1	c	70	8	2	0	80	15	.
PFW	*B. halimifolia*	(+)	2	a	57	12	6	5	80	42.5	0.093
PFW	*B. halimifolia*	(+)	2	b	66	13	1	0	80	18.75	.
PFW	*B. halimifolia*	(+)	2	c	53	14	9	4	80	49	.
PFW	*B. halimifolia*	(+)	3	a	78	2	0	0	80	2.5	0.301
PFW	*B. halimifolia*	(+)	3	b	67	11	2	0	80	18.75	.
PFW	*B. halimifolia*	(+)	3	c	75	4	0	1	80	7.5	.
PFW	*B. halimifolia*	(+)	3	d	73	7	0	0	80	8.75	.
PFW	*B. halimifolia*	(+)	3	e	78	2	0	0	80	2.5	.
PFW	*B. halimifolia*	(+)	3	f	80	0	0	0	80	0	.
PFW	*B. halimifolia*	(+)	3	g	58	7	5	10	80	46.25	.
PFW	*B. halimifolia*	(+)	3	h	77	3	0	0	80	3.75	.
PFW	*B. halimifolia*	(+)	3	i	64	10	5	1	80	27.5	.
PFW	*B. halimifolia*	(+)	4	a	62	11	4	3	80	31.25	0.573
PFW	*B. halimifolia*	(+)	4	b	72	7	1	0	80	11.25	.
PFW	*B. halimifolia*	(+)	4	c	71	6	3	0	80	15	.
PFW	*B. halimifolia*	(+)	4	d	73	5	2	0	80	11.25	.
PFW	*B. halimifolia*	(+)	4	e	80	0	0	0	80	0	.
PFW	*B. halimifolia*	(-)	1	a	77	3	0	0	80	3.75	0.235
PFW	*B. halimifolia*	(-)	1	b	75	3	2	0	80	8.75	.
PFW	*B. halimifolia*	(-)	1	c	74	5	1	0	80	8.75	.
PFW	*B. halimifolia*	(-)	1	d	72	6	2	0	80	12.5	.
PFW	*B. halimifolia*	(-)	1	e	80	0	0	0	80	0	.
PFW	*B. halimifolia*	(-)	1	f	73	7	0	0	80	8.75	.
PFW	*B. halimifolia*	(-)	2	a	80	0	0	0	80	0	0.39

Appendix D (I). Raw data from greenhouse experiment (Chapter 5), continued

PFW	*B. halimifolia*	(-)	2	b	72	8	0	0	80	12.5	.
PFW	*B. halimifolia*	(-)	2	c	78	0	2	0	80	2.5	.
PFW	*B. halimifolia*	(-)	2	d	74	6	0	0	80	7.5	.
PFW	*B. halimifolia*	(-)	2	e	75	5	0	0	80	6.25	.
PFW	*B. halimifolia*	(-)	2	f	76	4	0	0	80	5	.
PFW	*B. halimifolia*	(-)	3	a	80	0	0	0	80	1.25	0.039
PFW	*B. halimifolia*	(-)	3	b	70	9	1	0	80	12.5	.
PFW	*B. halimifolia*	(-)	3	c	80	0	0	0	80	0	.
PFW	*B. halimifolia*	(-)	3	d	80	0	0	0	80	0	.
PFW	*B. halimifolia*	(-)	3	e	80	0	0	0	80	5	.
PFW	*B. halimifolia*	(-)	3	f	69	7	4	0	80	25	.
PFW	*B. halimifolia*	(-)	4	a	62	9	5	4	80	28.75	0.087
PFW	*B. halimifolia*	(-)	4	b	63	12	3	2	80	22.5	.
PFW	*B. halimifolia*	(-)	4	c	78	1	0	1	80	11.25	.
PFW	*B. halimifolia*	(-)	4	d	63	10	4	3	80	21.25	.
PFW	*B. halimifolia*	(-)	4	e	78	2	0	0	80	5	.
PFW	*B. halimifolia*	(-)	4	f	75	3	0	2	80	6.25	.
PFW	*B. halimifolia*	(-)	4	g	80	0	0	0	80	0	.
PFW	*B. halimifolia*	(-)	4	h	80	0	0	0	80	15	.
PFW	*B. halimifolia*	(-)	4	i	59	9	5	7	80	40	.
PFW	*T. sebifera*	(+)	1	a	49	20	10	1	80	52.5	1.035
PFW	*T. sebifera*	(+)	1	b	51	19	8	2	80	48.75	.
PFW	*T. sebifera*	(+)	1	c	44	22	11	3	80	62.5	.
PFW	*T. sebifera*	(+)	1	d	58.5	15	4.5	2	80	35	.
PFW	*T. sebifera*	(+)	2	a	70	8	2	0	80	15	0.274
PFW	*T. sebifera*	(+)	2	b	56	15	6	3	80	41.25	.
PFW	*T. sebifera*	(+)	2	c	57	14	4	5	80	40	.
PFW	*T. sebifera*	(+)	2	d	38	29	11	2	80	68.75	.
PFW	*T. sebifera*	(+)	3	a	47	23	7	3	80	53.75	0.597
PFW	*T. sebifera*	(+)	3	b	68	12	0	0	80	15	.
PFW	*T. sebifera*	(+)	3	c	45	15	12	8	80	68.75	.
PFW	*T. sebifera*	(+)	3	d	68	7	4	1	80	22	.
PFW	*T. sebifera*	(+)	4	a	71	6	3	0	80	15	0.577
PFW	*T. sebifera*	(+)	4	b	60	13	7	0	80	33.75	.
PFW	*T. sebifera*	(+)	4	c	56	15	8	1	80	41.25	.
PFW	*T. sebifera*	(+)	4	d	52	13	7	8	80	53.75	.

Appendix D (I). Raw data from greenhouse experiment (Chapter 5), continued

PFW	*T. sebifera*	(+)	4	e	32	22	16	10	80	92.5	.
PFW	*T. sebifera*	(+)	4	f	27	23	18	12	80	103.75	.
PFW	*T. sebifera*	(-)	1	a	80	0	0	0	80	0	0.273
PFW	*T. sebifera*	(-)	1	b	76	4	0	0	80	5	.
PFW	*T. sebifera*	(-)	1	c	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	1	d	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	1	e	78	2	0	0	80	2.5	.
PFW	*T. sebifera*	(-)	1	f	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	1	g	77	3	0	0	80	3.75	.
PFW	*T. sebifera*	(-)	1	f	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	1	g	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	2	a	65	15	0	0	80	18.75	0.201
PFW	*T. sebifera*	(-)	2	b	77	3	0	0	80	3.75	.
PFW	*T. sebifera*	(-)	2	c	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	2	d	78	2	0	0	80	2.5	.
PFW	*T. sebifera*	(-)	2	e	76	4	0	0	80	5	.
PFW	*T. sebifera*	(-)	2	f	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	2	g	77	3	0	0	80	3.75	.
PFW	*T. sebifera*	(-)	2	f	78	2	0	0	80	2.5	.
PFW	*T. sebifera*	(-)	2	g	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	3	a	76	4	0	0	80	5	0.19
PFW	*T. sebifera*	(-)	3	b	78	2	0	0	80	2.5	.
PFW	*T. sebifera*	(-)	3	c	79	1	0	0	80	1.25	.
PFW	*T. sebifera*	(-)	3	d	78	2	0	0	80	2.5	.
PFW	*T. sebifera*	(-)	3	e	76	4	0	0	80	5	.
PFW	*T. sebifera*	(-)	3	f	77	3	0	0	80	3.75	.
PFW	*T. sebifera*	(-)	3	g	77	3	0	0	80	3.75	.
PFW	*T. sebifera*	(-)	3	f	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	3	g	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	4	a	80	0	0	0	80	0	0.199
PFW	*T. sebifera*	(-)	4	b	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	4	c	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	4	d	80	0	0	0	80	0	.
PFW	*T. sebifera*	(-)	4	e	76	4	0	0	80	5	.
PFW	*T. sebifera*	(-)	4	f	77	3	0	0	80	3.75	.
PFW	*T. sebifera*	(-)	4	g	77	3	0	0	80	10	.
PFW	*T. sebifera*	(-)	4	f	65	10	5	0	80	18.75	.

Appendix D (II). Raw data from field experiment (Harvest I) (Chapter 5). WPF = Wet pine forest, PFW = PFW, (+) = without fungicide, (–) = with fungicide, Rep = replication, % Colon = colonization

Habitats	Species	Rep	Sample	Negative score (VAM)	hyphae	arbuscule	vesicle	# Intersections	% Colon	Total biomass (g)
WPF	*B. halimifolia*	1	a	214	13	7	6	240	16.25	0.199
WPF	*B. halimifolia*	1	b	217	15	3	5	240	12.92	.
WPF	*B. halimifolia*	1	c	220	12	4	4	240	11.67	.
WPF	*B. halimifolia*	2	a	195	21	17	7	240	28.75	0.411
WPF	*B. halimifolia*	2	b	206	22	9	3	240	19.17	.
WPF	*B. halimifolia*	2	c	204	21	9	6	240	21.25	.
WPF	*B. halimifolia*	3	a	202	13	13	12	240	26.25	0.257
WPF	*B. halimifolia*	3	b	205	16	8	11	240	22.50	.
WPF	*B. halimifolia*	3	c	208	18	6	8	240	19.17	.
PFW	*B. halimifolia*	1	a	201	21	11	7	240	23.75	0.223
PFW	*B. halimifolia*	1	b	197	27	9	7	240	24.58	.
PFW	*B. halimifolia*	1	c	200	23	8	9	240	23.75	.
PFW	*B. halimifolia*	2	a	210	17	9	4	240	17.92	0.187
PFW	*B. halimifolia*	2	b	209	15	7	9	240	19.58	.
PFW	*B. halimifolia*	2	c	193	25	11	11	240	28.75	.
WPF	*M. cerifera*	1	a	223	11	3	3	240	9.58	0.052
WPF	*M. cerifera*	1	b	230	7	1	2	240	5.42	.
WPF	*M. cerifera*	1	c	234	5	0	1	240	2.92	.
WPF	*M. cerifera*	2	a	220	9	7	4	240	12.92	0.105
WPF	*M. cerifera*	2	b	221	11	5	3	240	11.25	.
WPF	*M. cerifera*	2	c	207	15	11	7	240	21.25	.
WPF	*M. cerifera*	3	a	224	7	5	4	240	10.42	0.006
WPF	*M. cerifera*	3	b	223	9	2	6	240	10.42	.
WPF	*M. cerifera*	3	c	223	12	3	2	240	9.17	.
PFW	*M. cerifera*	1	a	214	13	7	6	240	16.25	0.011
PFW	*M. cerifera*	1	b	230	4	2	4	240	6.67	.
PFW	*M. cerifera*	1	c	230	7	0	3	240	5.42	.
PFW	*M. cerifera*	2	a	215	15	7	3	240	14.58	0.095
PFW	*M. cerifera*	2	b	213	11	9	7	240	17.92	.
PFW	*M. cerifera*	2	c	217	14	4	5	240	13.33	.
PFW	*M. cerifera*	3	a	214	12	4	10	240	16.67	0.01
PFW	*M. cerifera*	3	b	228	8	3	1	240	6.67	.

Appendix D (II). Raw data from field experiment (Harvest I) (Chapter 5), continued

PFW	*M. cerifera*	3	c	220	10	5	5	240	12.50	.
WPF	*T. sebifera*	1	a	212	19	7	2	240	15.42	0.413
WPF	*T. sebifera*	1	b	194	23	11	12	240	28.75	.
WPF	*T. sebifera*	1	c	198	28	9	5	240	23.33	.
WPF	*T. sebifera*	2	a	183	33	14	10	240	33.75	0.638
WPF	*T. sebifera*	2	b	192	24	11	13	240	30.00	.
WPF	*T. sebifera*	2	c	196	27	13	4	240	25.42	.
WPF	*T. sebifera*	3	a	198	17	11	14	240	27.92	0.396
WPF	*T. sebifera*	3	b	199	21	15	5	240	25.42	.
WPF	*T. sebifera*	3	c	202	25	9	4	240	21.25	.
PFW	*T. sebifera*	1	a	193	31	13	3	240	26.25	0.311
PFW	*T. sebifera*	1	b	203	21	9	7	240	22.08	.
PFW	*T. sebifera*	1	c	195	23	11	11	240	27.92	.
PFW	*T. sebifera*	2	a	200	25	10	5	240	22.92	0.258
PFW	*T. sebifera*	2	b	199	23	9	9	240	24.58	.
PFW	*T. sebifera*	2	c	196	26	5	13	240	25.83	.
PFW	*T. sebifera*	3	a	197	28	7	8	240	24.17	0.457
PFW	*T. sebifera*	3	b	186	32	9	13	240	31.67	.
PFW	*T. sebifera*	3	c	183	37	13	7	240	32.08	.

Appendix D (III). Raw data from field experiment (Harvest II) (Chapter 5). WPF = Wet pine forest, PFW = PFW, (+) = without fungicide, (–) = with fungicide, Rep = replication, % Colon = colonization

Habitat	species	Rep	sample	Negative score (VAM)	hyphae	arbuscule	vesicle	# intersections	% Colon	Total biomass (g)
WPF	*B. halimifolia*	1	a	217	14	9	0	240	13.33	0.246
WPF	*B. halimifolia*	1	b	221	14	5	0	240	10.00	.
WPF	*B. halimifolia*	1	c	213	17	10	0	240	15.42	.
WPF	*B. halimifolia*	2	a	183	34	16	7	240	33.33	0.542
WPF	*B. halimifolia*	2	b	203	23	9	5	240	21.25	.
WPF	*B. halimifolia*	2	c	191	34	12	3	240	26.67	.
WPF	*B. halimifolia*	3	a	218	15	7	0	240	12.08	0.286
WPF	*B. halimifolia*	3	b	219	14	3	4	240	11.67	.
WPF	*B. halimifolia*	3	c	220	11	2	7	240	12.08	.
PFW	*B. halimifolia*	1	a	204	31	5	0	240	17.08	0.066
PFW	*B. halimifolia*	1	b	206	23	5	6	240	18.75	.
PFW	*B. halimifolia*	1	c	198	30	10	2	240	22.50	.
PFW	*B. halimifolia*	2	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	2	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	2	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	3	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	3	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	3	.	.	.	.	.	.	.	.
WPF	*M. cerifera*	1	a	225	9	2	4	240	8.75	0.159
WPF	*M. cerifera*	1	b	204	18	7	11	240	22.50	.
WPF	*M. cerifera*	1	c	225	11	3	1	240	7.92	.
WPF	*M. cerifera*	2	a	220	12	5	3	240	11.67	0.126
WPF	*M. cerifera*	2	b	227	10	2	1	240	6.67	.
WPF	*M. cerifera*	2	c	226	7	3	4	240	8.75	.
WPF	*M. cerifera*	3	a	237	3	0	0	240	1.25	0.101
WPF	*M. cerifera*	3	b	232	5	1	2	240	4.58	.
WPF	*M. cerifera*	3	c	229	7	1	3	240	6.25	.
PFW	*M. cerifera*	1	a	234	3	1	2	240	3.75	0.07
PFW	*M. cerifera*	1	b	227	10	1	2	240	6.67	.
PFW	*M. cerifera*	1	c	234	6	0	0	240	2.50	.
PFW	*M. cerifera*	2	a	221	13	3	3	240	10.42	0.067
PFW	*M. cerifera*	2	b	225	9	2	4	240	8.75	.

Appendix D (III). Raw data from field experiment (Harvest II) (Chapter 5), continued

PFW	*M. cerifera*	2	c	224	13	1	2	240	7.92	.
PFW	*M. cerifera*	3	.	.	.	.	.	.	.	.
PFW	*M. cerifera*	3	.	.	.	.	.	.	.	.
PFW	*M. cerifera*	3	.	.	.	.	.	.	.	.
WPF	*T. sebifera*	1	a	217	17	5	1	240	12.08	0.518
WPF	*T. sebifera*	1	b	214	16	7	3	240	15.00	.
WPF	*T. sebifera*	1	c	202	14	17	7	240	25.83	.
WPF	*T. sebifera*	2	a	202	28	5	5	240	20.00	0.893
WPF	*T. sebifera*	2	b	183	32	5	20	240	34.17	.
WPF	*T. sebifera*	2	c	199	29	9	3	240	22.08	.
WPF	*T. sebifera*	3	.	.	.	.	.	.	.	.
WPF	*T. sebifera*	3	.	.	.	.	.	.	.	.
WPF	*T. sebifera*	3	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	1	A	167	43	14	16	240	42.92	0.07
PFW	*T. sebifera*	1	B	161	33	24	22	240	52.08	.
PFW	*T. sebifera*	1	C	190	35	8	7	240	27.08	.
PFW	*T. sebifera*	2	A	187	28	18	7	240	32.50	0.067
PFW	*T. sebifera*	2	B	174	39	17	10	240	38.75	.
PFW	*T. sebifera*	2	C	177	32	17	14	240	39.17	.
PFW	*T. sebifera*	3	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	3	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	3	.	.	.	.	.	.	.	.

Appendix D (IV). Raw data from field experiment (Harvest III) (Chapter 5). WPF = Wet pine forest, PFW = PFW, (+) = without fungicide, (–) = with fungicide, Rep = replication, % Colon = colonization

Habitat	species	Rep	Sample	Negative score (VAM)	hyphae	arbuscule	vesicle	# Intersections	% Colon	Total biomass (g)
WPF	*B. halimifolia*	1	a	206	24	9	1	240	18.33	1.49
WPF	*B. halimifolia*	1	b	202	28	10	0	240	20.00	.
WPF	*B. halimifolia*	1	c	209	26	4	1	240	15.00	.
WPF	*B. halimifolia*	2	a	184	34	15	7	240	32.50	1.54
WPF	*B. halimifolia*	2	b	185	26	20	9	240	35.00	.
WPF	*B. halimifolia*	2	c	208	17	10	5	240	19.58	.
WPF	*B. halimifolia*	3	a	204	24	12	0	240	20.00	0.15
WPF	*B. halimifolia*	3	b	211	14	9	6	240	18.33	.
WPF	*B. halimifolia*	3	c	197	20	11	12	240	27.50	.
PFW	*B. halimifolia*	1	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	1	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	1	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	2	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	2	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	2	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	3	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	3	.	.	.	.	.	.	.	.
PFW	*B. halimifolia*	3	.	.	.	.	.	.	.	.
WPF	*M. cerifera*	1	a	198	21	11	10	240	26.25	0.09
WPF	*M. cerifera*	1	b	220	16	4	0	240	10.00	.
WPF	*M. cerifera*	1	a	226	13	1	0	240	6.25	0.08
WPF	*M. cerifera*	2	b	235	5	0	0	240	2.08	.
WPF	*M. cerifera*	2	.	.	.	.	.	.	.	.
WPF	*M. cerifera*	2	.	.	.	.	.	.	.	.
WPF	*M. cerifera*	3	.	.	.	.	.	.	.	.
WPF	*M. cerifera*	3	.	.	.	.	.	.	.	.
WPF	*M. cerifera*	3	.	.	.	.	.	.	.	.
PFW	*M. cerifera*	1	a	207	15	9	9	240	21.25	0.04
PFW	*M. cerifera*	1	b	206	18	10	6	240	20.83	.
PFW	*M. cerifera*	1	.	.	.	.	.	.	.	.
PFW	*M. cerifera*	2	.	.	.		.	.	.	.
PFW	*M. cerifera*	2	.	.	.		.	.	.	.

Appendix D (IV). Raw data from field experiment (Harvest III) (Chapter 5), continued

PFW	*M. cerifera*	2	.	.	.	.	.	.	.	.
WPF	*T. sebifera*	1	a	186	33	9	12	240	31.25	0.78
WPF	*T. sebifera*	1	b	182	24	21	13	240	38.33	.
WPF	*T. sebifera*	1	c	193	25	11	11	240	28.75	.
WPF	*T. sebifera*	2	a	187	27	12	14	240	32.92	0.49
WPF	*T. sebifera*	2	b	189	31	15	5	240	29.58	.
WPF	*T. sebifera*	2	c	201	20	10	9	240	24.17	.
WPF	*T. sebifera*	3	.	.	.	.	.	.	.	.
WPF	*T. sebifera*	3	.	.	.	.	.	.	.	.
WPF	*T. sebifera*	3	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	1	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	1	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	1	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	2	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	2	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	2	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	3	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	3	.	.	.	.	.	.	.	.
PFW	*T. sebifera*	3	.	.	.	.	.	.	.	.

Appendix D (V). Raw data from allelopathy experiment (Chapter 5). No Triadica = soil from Triadica uninvaded site, Triadica = soil from Triadica invaded site, (+) = with activated carbon, (–) = without activated carbon, Rep = replication, % Colon = colonization

Species	Soil source	Carbon (+/-)	Rep	Sample	Absence	Hyphae	Arbuscule	Vesicle	# Intersections	% Colon	shoot biomass (g)	root biomass (g)
B. halimifolia	No Triadica	(+)	1	a	208	21	7	4	240	17.92	4.237	1.01
B. halimifolia	No Triadica	(+)	1	b	228	11	1	0	240	5.42	.	.
B. halimifolia	No Triadica	(+)	2	a	199	24	8	9	240	24.17	5.239	0.74
B. halimifolia	No Triadica	(+)	2	b	217	16	2	5	240	12.50	.	.
B. halimifolia	No Triadica	(+)	3	a	215	11	5	9	240	16.25	3.372	0.87
B. halimifolia	No Triadica	(+)	3	b	222	13	5	0	240	9.58	.	.
B. halimifolia	No Triadica	(+)	4	a	226	11	2	1	240	7.08	3.754	0.62
B. halimifolia	No Triadica	(+)	4	b	232	8	0	0	240	3.33	.	.
B. halimifolia	No Triadica	(+)	5	a	222	11	3	4	240	10.42	2.942	0.49
B. halimifolia	No Triadica	(+)	5	b	189	33	7	11	240	28.75	.	.
B. halimifolia	No Triadica	(-)	1	a	204	24	12	0	240	20.00	1.786	0.34
B. halimifolia	No Triadica	(-)	1	b	209	25	6	0	240	15.42	.	.
B. halimifolia	No Triadica	(-)	2	a	202	28	5	5	240	20.00	2.667	0.28
B. halimifolia	No Triadica	(-)	2	b	175	32	17	16	240	40.83	.	.
B. halimifolia	No Triadica	(-)	3	a	207	17	9	7	240	20.42	0.11	0.02
B. halimifolia	No Triadica	(-)	3	b	226	9	3	2	240	7.92	.	.
B. halimifolia	No Triadica	(-)	4	a	210	11	7	12	240	20.42	2.757	0.31
B. halimifolia	No Triadica	(-)	4	b	230	6	3	1	240	5.83	.	.
B. halimifolia	No Triadica	(-)	5	a	196	31	11	2	240	23.75	3.881	0.36
B. halimifolia	No Triadica	(-)	5	b	216	17	3	4	240	12.92	.	.
B. halimifolia	Triadica	(+)	1	a	214	18	7	1	240	14.17	2.739	0.73
B. halimifolia	Triadica	(+)	1	b	207	28	3	2	240	15.83		
B. halimifolia	Triadica	(+)	2	a	220	10	7	3	240	12.50	3.423	1.32
B. halimifolia	Triadica	(+)	2	b	210	20	9	1	240	16.67		
B. halimifolia	Triadica	(+)	3	a	218	19	3	0	240	10.42	3.77	1.23
B. halimifolia	Triadica	(+)	3	b	218	16	5	1	240	11.67	.	.
B. halimifolia	Triadica	(+)	4	a	239	1	0	0	240	0.42	3.364	1.05
B. halimifolia	Triadica	(+)	4	b	226	11	1	2	240	7.08	.	.
B. halimifolia	Triadica	(+)	5	a	226	11	2	1	240	7.08	3.103	1.51
B. halimifolia	Triadica	(+)	5	b	227	12	1	0	240	5.83	.	.
B. halimifolia	Triadica	(-)	1	a	219	18	3	0	240	10.00	2.84	0.71

Appendix D (V). Raw data from allelopathy experiment (Chapter 5), continued

B. halimifolia	Triadica	(-)	1	b	210	19	6	5	240	17.08	.	.
B. halimifolia	Triadica	(-)	2	a	228	11	1	0	240	5.42	1.97	0.48
B. halimifolia	Triadica	(-)	2	b	214	17	6	3	240	14.58	.	.
B. halimifolia	Triadica	(-)	3	a	203	22	15	0	240	21.67	2.948	0.51
B. halimifolia	Triadica	(-)	3	b	221	14	5	0	240	10.00	.	.
B. halimifolia	Triadica	(-)	4	a	203	25	9	3	240	20.42	2.436	0.6
B. halimifolia	Triadica	(-)	4	b	211	25	2	2	240	13.75	.	.
B. halimifolia	Triadica	(-)	5	a	221	15	3	1	240	9.58	2.866	1.45
B. halimifolia	Triadica	(-)	5	b	211	18	7	4	240	16.67	.	.
M. cerifera	No Triadica	(+)	1	a	233	5	2	0	240	3.75	2.018	0.82
M. cerifera	No Triadica	(+)	1	b	218	8	7	7	240	15.00	.	.
M. cerifera	No Triadica	(+)	2	a	236	3	1	0	240	2.08	2.267	1.33
M. cerifera	No Triadica	(+)	2	b	240	0	0	0	240	0.00	.	.
M. cerifera	No Triadica	(+)	3	a	232	5	3	0	240	4.58	2.854	1.35
M. cerifera	No Triadica	(+)	3	b	228	7	5	0	240	7.08	.	.
M. cerifera	No Triadica	(+)	4	a	230	6	4	0	240	5.83	1.189	0.51
M. cerifera	No Triadica	(+)	4	b	237	3	0	0	240	1.25	.	.
M. cerifera	No Triadica	(+)	5	a	222	10	5	3	240	10.83	1.339	0.57
M. cerifera	No Triadica	(+)	5	b	239	1	0	0	240	0.42	.	.
M. cerifera	No Triadica	(-)	1	a	229	6	5	0	240	6.67	2.312	0.26
M. cerifera	No Triadica	(-)	1	b	232	7	1	0	240	3.75	.	.
M. cerifera	No Triadica	(-)	2	a	202	20	12	6	240	23.33	1.455	1.1
M. cerifera	No Triadica	(-)	2	b	213	18	5	4	240	15.00	.	.
M. cerifera	No Triadica	(-)	3	a	222	13	4	1	240	9.58	1.362	0.6
M. cerifera	No Triadica	(-)	3	b	218	14	4	4	240	12.50	.	.
M. cerifera	No Triadica	(-)	4	a	231	7	2	0	240	4.58	2.335	1.14
M. cerifera	No Triadica	(-)	4	b	231	5	2	2	240	5.42	.	.
M. cerifera	No Triadica	(-)	5	a	240	0	0	0	240	0.00	0.876	0.29
M. cerifera	No Triadica	(-)	5	b	240	0	0	0	240	0.00	.	.
M. cerifera	Triadica	(+)	1	a	234	2	2	2	240	4.17	0.867	0.25
M. cerifera	Triadica	(+)	1	b	235	3	2	0	240	2.92	.	.
M. cerifera	Triadica	(+)	2	a	232	5	3	0	240	4.58	1.998	0.59
M. cerifera	Triadica	(+)	2	b	236	2	2	0	240	2.50	.	.
M. cerifera	Triadica	(+)	3	a	233	6	1	0	240	3.33	1.045	0.27
M. cerifera	Triadica	(+)	3	b	236	4	0	0	240	1.67	.	.
M. cerifera	Triadica	(+)	4	a	228	10	2	0	240	5.83	1.777	0.45

Appendix D (V). Raw data from allelopathy experiment (Chapter 5), continued

M. cerifera	Triadica	(+)	4	b	234	4	1	1	240	3.33	.	.
M. cerifera	Triadica	(+)	5	a	227	8	3	2	240	7.50	4.065	1.58
M. cerifera	Triadica	(+)	5	b	233	5	2	0	240	3.75	.	.
M. cerifera	Triadica	(-)	1	a	232	5	3	0	240	4.58	0.051	0.33
M. cerifera	Triadica	(-)	1	b	235	3	2	0	240	2.92	.	.
M. cerifera	Triadica	(-)	2	a	223	11	4	2	240	9.58	1.471	0.54
M. cerifera	Triadica	(-)	2	b	236	2	1	1	240	2.50	.	.
M. cerifera	Triadica	(-)	3	a	235	5	0	0	240	2.08	1.754	0.65
M. cerifera	Triadica	(-)	3	b	232	7	0	1	240	3.75	.	.
M. cerifera	Triadica	(-)	4	a	226	7	3	4	240	8.75	0.568	0.23
M. cerifera	Triadica	(-)	4	b	234	3	2	1	240	3.75	.	.
M. cerifera	Triadica	(-)	5	a	227	6	4	3	240	8.33	1.427	0.69
M. cerifera	Triadica	(-)	5	b	236	3	1	0	240	2.08	.	.

VITA

Graduate School

Southern Illinois University

Shishir Paudel

paudelshishir@gmail.com

Tribhuvan University, Kathmandu, Nepal
Masters of Science in Botany, November 2001

University of Bergen, Norway
Master of Science in Biology, January 2006

Special Honors and Awards:
Study abroad program, The Norwegian Educational Fund, January 2004 – December 2005

Dissertation Title:
Climate change, shifts in tropical storm regimes and *Triadica sebifera* invasion in coastal Mississippi, United States

Major Professor: Dr. Loretta L. Battaglia

Publications:

1. **Paudel, S.** and LL. Battaglia. Germination responses of the invasive *Triadica sebifera* and two co-occurring native woody species to elevated salinity across a Gulf Coast transition ecosystem. Wetlands (**Accepted**).
2. **Paudel, S.** & J. Sah, 2003. Physiochemical Characteristics of Soil in tropical Sal (*Shorea robusta*) Forests in Eastern Nepal. *Himalayan Journal of Sciences*, **1**: 107-110.
3. **Paudel, S.** 2003. Community Forestry in Nepal: An Overview. *Himalayan Journal of Sciences*, **1**: 62-65.
4. Shakya, P.R., I. Shrestha, **S. Paudel** & S. Shakya. 2002. Study on non-timber forest product in Annapurna Conservation Area (ACA), Nepal. *Annual Report of Pro. Natura Fund*, **11**:109-132.

Abstracts:

1. **Paudel, S.** and L.L. Battaglia. Resilience of Floating Aquatic Plant Communities in Coastal Bottomlands of the Lower Mississippi Alluvial Valley (LMAV). Oral presentation. Annual meeting of the Ecological Society of America meeting, Portland, OR, August 2012. National Meeting.
2. **Paudel, S.** and L. L. Battaglia. Growth response of native and invasive species within coastal transition ecosystems at Grand Bay, Mississippi, USA: the effect of canopy and storm surge. Oral presentation. Annual meeting of the Ecological Society of America meeting, Austin, TX, August 2011. National Meeting.
3. **Paudel, S.** and L.L. Battaglia. Resilience of Floating Aquatic Plant Communities in Coastal Bottomlands of the Lower Mississippi Alluvial Valley (LMAV). Oral presentation. Annual intern symposium of the National Great Rivers Research and Education Center (NGRREC), Alton, IL, August 2011.
4. **Paudel, S.** and L.L. Battaglia. Predicting effects of altered tropical storm regimes on recruitment of native and invasive species along the coastal transition ecosystem at Grand Bay, Mississippi, USA. Oral presentation. Annual meeting of the Ecological Society of America, Pittsburgh, PA, August 2010. National Meeting.
5. **Paudel, S.** and L.L. Battaglia. Climate change and potential shifts in distribution of exotic and native plant species in Coastal Ecosystems of the Northern Gulf of Mexico. Oral presentation. Annual meeting of the Society for Wetland Scientists, Madison, Wisconsin, June 2009. International Meeting.
6. **Paudel, S.** and L.L. Battaglia. Predicting the invasiveness of an alien plant *Triadica sebifera* under future climate change scenarios in southeastern US. Oral presentation. Annual meeting of The Illinois Academy of Sciences. Southern Illinois University Edwardsville, April 2009. Regional Meeting.
7. Makweche, P.T, **S. Paudel,** J. Looft, G. D. Schmale, D. R. Ernat and L. L. Battaglia. Fire suppression effects on a coastal floating marsh ecosystem. Poster presentation. Annual meeting of the Society for Wetland Scientists, Washington, D. C., May 2008. International meeting.