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NATURAL REGENERATION DYNAMICS AND SURVIVAL INFLUENCED BY ABIOTIC AND BIOTIC FACTORS IN A BOTTOMLAND HARDWOOD FOREST

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By

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ABSTRACT

Riparian ecosystems are vital to the landscape, providing critical services including water filtration and purification, flood and erosion control, carbon sequestration, biodiversity support, and aesthetic value. Bottomland hardwood forests, however, are threatened by invasive species, land loss/conversion, inconsistent or absence of harvesting disturbances, and altered hydrological patterns, leading to reduced success of desired, native species. This research assessed regeneration dynamics and one-year survival in a seasonally-flooded bottomland hardwood forest at Boggy Slough Conservation Area in East Texas to identify abiotic and biotic factors important for successful establishment of native regeneration. Areas sampled included two that were previously treated with herbicide for Chinese tallow (*Triadica sebifera* L.) and one untreated, reference area.

Shade-tolerant species were, in general, more common in the regeneration layer than shade-intolerant species. Less-desirable native species, such as water-elm (*Planera aquatica* J.F.Gmel.) and American hornbeam (*Carpinus caroliniana* Walt.) as well as invasive tallow were present at higher densities than many desired species. Oak species had low sapling densities, indicating potential recruitment issues throughout much of the site. The results showed that seedling densities of some native species, as well as Chinese tallow, varied among treatment areas, and that new cohorts of tallow regeneration may have been facilitated in the post-treatment environment at the expense of certain native species. Overcup oak (*Quercus lyrata* Walt.) seedlings were most abundant in the reference area. Other native indicator species in both treated and reference areas were mid-tolerant to tolerant of shade. Abiotic and biotic factors, representing complex moisture, light, and ground cover gradients, were associated with regeneration abundances of native species differently across sapling (DBH 0.6 to 3.9 inches), large seedling (DBH < 0.6 inches, height ≥ 6 inches), and small seedling (DBH < 0.6 inches, height < 6 inches) size classes.

After one year, survival rates among native-hypogeal, native-epigeal, and invasive-Chinese tallow differed; Chinese tallow had a significantly lower survival rate than the native groups, likely as a result of weather events. Height and diameter increased the odds of survival for all regeneration groups. Additionally, vigor, down woody debris (DWD), microtopography, and canopy cover had varying influence among species groups. When significant, raised microsites increased odds of survival, while proximity to DWD and higher canopy cover decreased odds of survival. This research provided deeper insight into micro-site preferences of bottomland native tree species and which are projected to occupy the canopy in the future. To encourage regeneration of desired, native species, managers must be able to match species to site and create canopy openings sufficient to promote recruitment of native species without allowing competitive invasive species from claiming newly-available light and growing space.

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INTRODUCTION

Riparian forests, commonly referred to as floodplain, lowland, or bottomland hardwood forests, are vital ecosystems in parts of the United States and around the globe. Floodplain forests, located alongside meandering streams and rivers, are unique, transitional zones between permanently flooded wetlands and upland forests. These forests experience flood regimes that vary in duration from seasonal to semi-permanent flooding based on subtle heterogeneity in topography within the river floodplain (Collins & Battaglia 2008). Flooding is an essential disturbance to these ecosystems, influencing geomorphology, soil moisture, and seed dispersal (Kroschel et al. 2016, Romano 2010). Natural flooding also brings sediment into the forest, providing new substrate for seedling establishment (Kozlowski 2002) and influencing succession (Hodges 1997). Vegetation in these ecosystems is significantly influenced by environmental gradients, particularly moisture (Mitch & Gosselink 2015). The array of benefits from riparian ecosystems has not been consistently understood. Currently, many of the world's floodplain forests are declining or in a degraded state due to numerous human effects and subsequent lack of desired species' regeneration. To ensure the future of these wetland forests, research regarding the regeneration of native tree species is critical in the face of increasing pressure from anthropogenic changes and biological invasions.

In 1987, the U.S. Army Corps of Engineers and the Environmental Protection Agency provided an updated definition of wetlands founded on the presence of specific hydrological, edaphic, and vegetational traits. This characterization allowed large expanses of the country's riparian forests to be classified as wetlands, giving them federal protection under the Clean Water Act of 1977 (Kellison & Young 1997). Based on their position in the landscape, riparian forests provide a wide range of ecosystem services (Dybala et al. 2019, Kellison & Young 1997, Kozlowski 2002, Lowrance et al. 1984, Schindler et al. 2014). They play an important role in water quality enhancement, nutrient cycling, and mitigation of pollution. Acting as natural reservoirs, they fill with upstream precipitation and runoff, and nutrients are then deposited with sediments, improving water quality downstream. This is especially important in agricultural watersheds, where nutrient runoff levels are high. Riparian forests have been called the landscape's "kidneys" (Kellison & Young 1997), as they convert chemical waste and toxins into harmless forms of biomass. The soils, with their high amounts of organic matter, regular waterlogging, and large inputs of nitrate through subsurface flow, provide optimal conditions for denitrification (Kellison & Young 1997, Kozlowski 2002, Lowrance et al. 1984). Additionally, these ecosystems provide flood and erosion control by holding water and sediment (Kellison & Young 1997, Kozlowski 2002). Due to their heterogeneity of microsites and frequency of disturbance, they are significant sources of biodiversity for flora and fauna. They make excellent habitat for wildlife and are often prized for their aesthetics and recreational opportunities (Kellison & Young 1997, Kozlowski 2002, Schindler et al. 2014). Healthy, functioning floodplain forests also have the ability to be significant carbon sinks, giving them potential to help mitigate the ever-growing effects of climate change (Dybala et al. 2019). These ecosystem services make riparian forests highly valuable and of importance for conservation efforts.

While lowland hardwood forests are valuable, they are also vulnerable. Conversion to agricultural fields or urban developments has been a leading cause of imperiling forested wetlands (Dybala et al. 2019). As of the end of the 20th century, over half of the pre-colonial wetlands in the continental United States had been lost. The southeastern region of the United States, home to an abundance of bottomland hardwood (BLHW) ecosystems, has been especially impacted. A large proportion of BLHW forests were lost to deforestation and conversion in the 1960s and 70s, as these lands provided productive soils for crops (Kellison & Young 1997). In the Lower Mississippi Alluvial Valley, an estimated 80% of the BLHW forests have been converted to agricultural crop land. Similarly, East Texas has lost approximately 60% of its floodplain forests (Kroschel et al. 2016). With such a large proportion lost, the health and productivity of the remaining riparian forests must be protected. Additionally, historical patterns of high-grading or lack of proper stand management have left many remaining BLHW forests in a degraded state (Kellison & Young 1997).

Altered hydrology and geomorphology threaten almost all floodplain forest ecosystems. Beginning in the late 1800s, river alterations such as dams and levees became common for purposes such as navigation, flood regulation, and storage reservoirs. There are commonly storage reservoirs in headwater regions of regulated rivers, and one reservoir has the potential to affect the downstream flow for nearly the entire extent of a river (Dybala et al. 2019, Kroschel et al. 2016, Nilsson & Berggren 2000). As each river and associated dam is unique, so are the corresponding effects to the upstream and downstream ecosystems (Katz et al. 2005, Nilsson & Berggren 2000, Saladyga et al. 2020). However, the disruption of natural flooding regimes elicits some shared issues, though they may vary in extent. Factors such as timing, frequency, intensity, and spatial extent of flooding are altered by river regulation. Changing hydrological patterns leads to modified amounts and movements of water, sediment, nutrients, and organic matter. Dams specifically often reduce downstream peak flooding and flooding frequency or temporally displace them. Groundwater levels can also be lowered (Kozlowski 2002, Nilsson & Berggren 2000). These and other complex changes can result in hindered plant growth and regeneration, as well as increased mortality rates. Over time, these negative effects to the plant communities lead to a decrease in plant establishment and subsequent species composition changes. Riparian pioneer species decline as their ideal conditions are lost. In many BLHW forests, there are observed shifts from more shade-intolerant and flood-tolerant species toward shade-tolerant and less flood-tolerant species. Ultimately, a lack of desired regeneration adversely directs plant succession to older and less productive conditions (Kozlowski 2002, Kroschel et al. 2016, Nilsson & Berggren 2000).

Along with land loss and hydrologic alterations, invasions by non-native species also pose a serious threat to native regeneration. Floodplain forests are sensitive ecosystems and are one of the most invaded forest environments. Even when unregulated, riparian forests are naturally susceptible to the incursion of exotic plant species due to their regular disturbances, high availability of water, and widespread dispersal system through rivers. Longitudinal hydraulic connectivity provides opportunities for seeds to disperse downstream, while lateral water movement changes soil nutrients, exposes substrate, and stresses the already present plants. These characteristics provide opportunities for colonization and establishment for both native and non-native seeds. The vulnerability of floodplains can become more drastic in human-altered ecosystems; the modified environment may suit exotic species better than native species or cause the native, flood tolerant species to lose their competitive advantage (Katz et al. 2005, Nilsson & Berggren 2000, Predick & Turner 2008, Repka et al. 2015). The success of exotic invasion depends on the habitat quality and the competition posed by current species. Habitat quality is determined by factors such as soil, land-use history, and topography. Sites with high nutrient levels, low vegetation density, and abundant light are commonly victims of invasion (Predick & Turner 2008).

The establishment of invasive species threatens natural biodiversity and ecosystem functions, affects natural disturbance regimes, and has been correlated with growth and survival declines in native plant communities (Greene & Blossey 2012, Predick & Turner 2008). Woody invasive plants are an exceptional menace because they can grow to negatively affect several strata of a forest community, including seedlings and saplings. Young seedlings are highly vulnerable to stressors, including competition, which can significantly impact future forest structure and composition. Early successional native riparian species can be outcompeted by invasive exotics, leading to a decline in regeneration rates (Boyce 2015, Predick & Turner 2008, Romano 2010, Terwei et al. 2013, Webster et al. 2006). Thus, consideration of the effects that exotic species and treatments to manage them have on regeneration dynamics is important when studying invaded lowland forests.

Invaluable ecosystems, riparian forests are threatened and in need of restoration and careful management. Understanding the complex ecology of these forests is fundamental to determining silvicultural tools that will best facilitate the next cohorts of native species. This study identified the influence of biotic and abiotic factors on native tree regeneration in floodplain forests. The BLHW forests of the southern United States were the specific focus of this research project. Most of these forests are negatively affected by modern hydrological changes and the introduction of invasive species such as Chinese tallow (*Triadica sebifera* L.). Novel research is needed to analyze how bottomland forests are impacted by the occurrence and treatment of tallow. Examination of current natural regeneration patterns of native and non-native tree species can aid managers in further understanding the current status of the ecosystem and pinpointing restoration needs. Additionally, evaluating how past treatments for invasive species may have affected the remaining flora and environment further aided this goal.

The overall goal of this research was to identify where natural regeneration of native tree species has been successful in establishing in bottomland hardwood forests adjacent to river systems. The objectives in Chapter II were to 1) quantify current regeneration dynamics (species composition, density, and height and diameter of established seedlings and saplings) in areas previously treated with herbicide to reduce Chinese tallow occurrence and untreated, reference areas and 2) evaluate the influence of abiotic and biotic factors (e.g., microtopography; down woody debris mass; overstory density; canopy cover; ground cover by leaf litter, grasses, forbs, and shrubs; density of Chinese tallow seedlings and saplings; occurrence of feral hog damage) on the occurrence of native seedlings and saplings. The objectives in Chapter III were to 1) compare one-year survival (2020-2021) rates among native hypogeal, native epigeal, and invasive Chinese tallow species groups for selected small and large seedlings and saplings and 2) determine the influence of silvical characteristics, stem size (height and diameter) and vigor, canopy cover, and micro-location (microtopography, proximity to DWD) on one-year survival native hypogeal, native epigeal, and invasive Chinese tallow seedlings and saplings.

LITERATURE REVIEW

ECOLOGY OF BOTTOMLAND HARDWOOD FORESTS

FLOODPLAIN LANDSCAPE AND VEGETATION PATTERNS

Bottomland hardwood forests are mesic riparian ecosystems that commonly occur adjacent to meandering streams or rivers capable of producing overland flow. This forest type occurs throughout the central and eastern United States and is of particular influence in the southeastern region. BLHW forests are found in zones of deposition that are influenced by river pulses in the winter and spring and drier conditions for at least part of the growing season (Mitch & Gosselink 2015). Due to this, the soils show minimal profile development, with texture dependent on fluvial landforms and distance from the channel (Hodges 1997).

Distinct geomorphic features characterize broad alluvial plains due to the extensive movement of the river channel eroding, moving, and depositing alluvium. Natural levees are embankments that border the river channel, composed of coarse sand particles that are deposited first during flood events. Point bars occur on the convex side of river curves where sediments are deposited. Over long periods of time, these form meander scrolls, which are ridges and depressions remaining after the river migrates laterally across a floodplain. Sloughs can develop in these meander scrolls and collect standing water. Oxbow lakes form in areas where the channel formerly meandered, with water remaining in them due to clay plugs. Floodwater deposits finer clay sediments at the furthest points of their extension; this is where backswamps develop. Higher in the valley, abandoned floodplains known as terraces occur. Terraces have older soils, are seldom flooded, and are less productive than active floodplains (Hodges 1997, Mitch & Gosselink 2015).

Fluvial landforms are important influencers of vegetation patterns in floodplain forests, as they provide varying heights above the river channel and thus dictate soil moisture content and oxygen availability (Hupp & Osterkamp 1985, Romano 2010). The tree species found in BLHW forests vary in flood tolerance along these environmental gradients created by micro-relief, which determines the depth and duration of inundation. Natural levees and ridges experience shorter hydroperiods and thus contain less flood tolerant species. Lower areas, such as sloughs, oxbow lakes, and backswamps are saturated for longer periods and thus contain more flood tolerant species. Many floodplain species are mesophytic and are only moderately tolerant of flooding (Gravatt & Kirby 1997, Kozlowski 2002, Mitch & Gosselink 2015). Flood tolerance is influenced by factors such as the species and age of a tree, as well as the length of time it is inundated. Extended flooding events can kill small, poor vigor, or immature trees, resulting in a more even-sized structure amongst the older, healthier survivors. Woody plants found in floodplain environments have adaptations such as aerenchyma, hypertrophied lenticels, and adventitious roots to survive periods of anaerobic soils. Along with flooding regimes, composition is further influenced by abiotic factors such as soil, fire, and timber harvesting and biotic factors such as shade tolerances and competition for light within and among species (Romano 2010).

FOREST SUCCESSION AND REGENERATION

Secondary succession in riparian forests is both autogenic and allogenic, influenced by past disturbances, light availability and gap sizes, soil characteristics, drainage, local elevation, and rate of deposition. Species that occur within each stage is dependent on the particular region. Early-seral species in southern BLHW forests include black willow (*Salix nigra* Marsh.) in wetter areas and eastern cottonwood (*Populus deltoides* Bartr. ex Marsh.) in better-drained areas. Deposition may lead to a progression towards sub-climax communities such as elm-ash-hackberry or red oak-sweetgum, which can self-replace and also last for hundreds of years if site conditions remain similar. When the deposition rate slows or halts, a climax type of oak-hickory will eventually develop and persist. On low elevations sites with slow deposition, succession can progress to an overcup oak-water hickory type. Altered river hydrology is also a factor influencing succession; forest composition can shift to less flood-tolerant, mesic species (Hodges 1997, Kroschel et al. 2016).

Trees reproduce sexually through seed production, dispersal, germination, establishment, and survival. Species can be characterized by heavy- (e.g., oaks, hickories) or light-seeded (e.g., elms, maples, ashes) reproductive strategies. Heavy-seeded species invest in larger seeds with higher rates of seed survival but lower yields, while light-seeded species devote resources to large amounts of small seeds with high mortality rates. Flooding is a key mechanism for seed dispersal in the spring and early summer. Timing of flooding is important for some species, as they require certain conditions to succeed in the floodplain environment (Kroschel et al. 2016, Romano 2010). Many species, especially bottomland oaks, rely on water to propel or roll seeds across the forest floor. Overcup oak (*Quercus lyrata* Walt.) has seeds that can float which aids its dispersal. Other key dispersal mechanisms are animals (e.g., birds and small mammals), wind, and gravity (Kroschel et al. 2016, Streng et al. 1989).

Germination occurs in a window of opportunity where the conditions must be favorable and meet the life history requirements of a species. Epigeal germination is exhibited in species whose cotyledons grow about the soil surface, whereas the cotyledon remains below ground in species whose germination is hypogeal (Copeland & McDonald 1999). Some species, such as overcup oak, may benefit from longer periods of flooding, while this may hinder other oak species. Seeds that germinate earlier have the advantage of a longer growing season, but are also exposed to more flooding disturbance. Light-seeded species tend to germinate earlier in the growing season, benefiting from higher light levels due to the open canopy. Heavy-seeded species emerge later, relying on the endosperm for survival; that extra energy reserve likely compensates for a shorter growing season (Kroschel et al. 2016, Streng et al. 1989).

Kroschel et al. (2016) define establishment as the period from emergence through the first three growing seasons, and survival as the time beyond this until mortality. The probability of establishment depends on light availability, microtopography, and the seasonality of emergence. Flooding stress lowers the ability of seedlings to photosynthesize, and the establishment stage of bottomland seedlings can be vulnerable with a high mortality rate. Seedling survival is affected by factors such as precipitation, competition, and the water-table level. As juveniles mature into the sapling stage and later mature stage, the importance of light grows while the impact of flooding diminishes. Canopy gaps and drought/flood events facilitate species diversity. Gaps increase light penetration to the forest floor, releasing less-shade-tolerant oaks (Kroschel et al. 2016).

INFLUENCES ON REGENERATION DYNAMICS AND SURVIVAL IN RIPARIAN FORESTS

ABIOTIC FACTORS

Light and flooding (hydroperiod) are the most influential environmental variables on dynamics and establishment of regeneration in floodplain forests (Battaglia et al. 2000, Battaglia & Sharitz 2006, Hall & Harcombe 1998, Kroschel et al. 2016, Küßner 2003). One study found that the density of seedlings over a year old was correlated with light availability in the summer (Streng et al. 1989). Battaglia et al. (2000) found in a simulated study that light and water can affect regeneration both individually and through interaction. Water levels were influenced by microtopography. The emergence of swamp chestnut oak (*Quercus michauxii* Nutt.) seedlings was higher on mounds at all light levels, but significantly lower in pits at the lowest light levels. Sweetgum (*Liquidambar styraciflua* L.) had similar results on mounds, but in full sunlight emergence was reduced. This may have been due to a combined stress from increased heat and decreased moisture. In both species, mortality was affected by water-table depth but not light, with higher rates at lower elevations and lower rates at higher elevations. The flood-light interaction was also found in a floodplain forest in Germany; pedunculate oak (*Quercus robur* L.) was impacted to a greater degree by low light levels at higher elevations, perhaps because the seedlings could not compensate for the lack of light with high water availability (Küßner 2003). In a field study in South Carolina, Battaglia and Sharitz (2006) discovered that the water table depth alone did not significantly affect sapling species, but when light was included, less shade-tolerant species were found in open, wetter areas while shade-tolerant species tended to occur on drier sites. The sediment load in the water can reduce the amount of light reaching the seedlings if they are submerged and could settle on leaf surfaces when the water recedes. In a greenhouse study, shaded silver maple (*Acer saccharinum* L.) seedlings submersed in cloudy water had much lower photosynthetic rates than their counterparts in clear water (Peterson & Bazzaz 1984).

At the microsite level, seedlings at lower elevations experience longer flooding duration than those higher up. In Peterson & Bazzaz (1984), the duration of submersion more strongly impacted the physiological functions of seedlings than the light condition they were growing in, with a decrease in net photosynthesis as flooding time increased. In an East Texas study, the variation in flooding response in light seeded species was attributed to microsite-level differences in elevation. Elm and maple, for example, had higher mortality levels during high flooding in lower areas, while American hophornbeam (*Ostrya virginiana* (Mill.) K. Koch) and holly (*Ilex decidua* Walt.) in the highest microsites were not affected by flooding. Flooding variation was determined to be the likely highest influencer of overall numbers of seedlings (i.e., density), and results indicated that density of light-seeded species was reduced by flooding, while heavy-seeded species were less impacted. Alongside the lack of oxygen, damage to seedlings by floating debris may also be a leading cause of regeneration mortality (Streng et al. 1989). Hall & Harcombe (2001) found that, over multiple years, flooding intensity was correlated with mortality of saplings of various size classes. They concluded that canopy gaps in combination with drought and flood events created conditions conducive to species diversity.

BIOTIC FACTORS

Little research has been performed on how organic matter depth may affect regeneration in floodplain forests. Plant litter has the potential to affect the germination and establishment of seedlings. Litter can aid by enhancing the soil moisture, reducing competition, and protecting seeds from predation. However, litter can be detrimental to regeneration by reducing temperature or burying, obstructing, and shading seeds. Plant litter can increase soil nutrients by releasing them during decomposition. Decomposition rates tend to be higher in riparian corridors because of increased decomposer activity and higher leaching rates. River-born litter may also improve nutrient uptake by trapping suspended materials in the water. Compared to uplands, litter may be less impactful in riparian systems because it is not present for as long (Xiong & Nilsson 1997). Additionally, there are little data regarding how proximity to downed woody debris could impact regeneration survival in these systems. In some wetlands, tree seedlings that grow on large pieces of woody debris have a greater chance of survival and success. The position can elevate the seedling above other ground cover forms to receive more light or provide less saturated mediums for germination. Regeneration near woody debris could benefit from protection from herbivory and burial as well as altered microsites that could provide a less stressful growing environment (Harmon et al. 1986, Sharitz 1996, Stevens 1997).

Ground cover competition by tall herbaceous flora can have an impact on tree regeneration. A study on the Upper Rhine River found that shading below 5-10 percent reduced tall herbaceous cover, allowing woody species to establish. Moderate shading and flooding that reduce competitive herbaceous cover may enhance the survival probability of slow-growing tree seedlings (Siebel & Bouwma 1998). Competition from invasive plants, in particular, can have an influence on the regeneration dynamics of native species. Alien trees affect multiple strata in forests, competing for growing space with trees in the canopy, as well as outcompeting native juveniles when gaps open

(Webster et al. 2006). When invasive seedlings establish, they can reduce regeneration rates and impede recruitment of native species. Additionally, when exotic trees mature, they may facilitate the success of their own seedlings rather than that of native species. In a riparian forest in northern Italy, invasive black cherry (*Prunus serotina* Ehrh.) had a negative effect on native field elm (*Ulmus minor* Mill.) regeneration, but invasive black locust (*Robina pseudoacacia* L.) facilitated native pedunculate oak vegetation (Terwei et al. 2013). Reinhart et al. (2005) investigated the effects of Norway maple (*Acer platanoides* L.) invasion in a riparian area and observed that the regeneration of indigenous canopy trees was suppressed by the invasive tree. Part of this effect was attributed to a reduction of light intensity in the understory, hindering the native species. They also noted that the invaded stands had a more homogenous species composition. Similarly, Gorchov and Tisel (2003) found that aboveground competition with Amur honeysuckle (*Lonicera maackii* (Rupr.) Maxim.) reduced the density of native woody seedlings in invaded forests. In a potted plant experiment, invasive reed canary grass (*Phalaris arundinacea* L.) decreased the size and survival of silver maple seedlings (Thomsen et al. 2012).

Feral hogs (*Sus scrofa* L.) are a prominent threat to the bottomland forests of the southern United States. As they "root" for food, they disrupt the soil and litter layers and cause mechanical damage to existing plants. One study found that rooting from wild pigs damaged over 25% of a floodplain forest floor (Jones et al. 1994). Rooting can influence soil structure and chemistry by mixing horizons, reducing litter, and increasing leaching of plant nutrients (Singer et al. 2984). Along with disturbing the seed bed, feral hogs feed on both hard and soft mast, and favor young, green seedlings of trees and herbs. They also may prefer oak and hickory fruits, which can alter amounts and distribution of those species in the future (Kroschel et al. 2016, Wood & Roark 1980). Disruption of the soil by hogs can change the chemistry of the forest floor and reduce diversity in the regeneration layer. Another study found that locations without hog enclosures had twice the abundance of Chinese tallow (Pile et al. 2017, Siemann et al. 2009).

Individual seedling survival is also influenced by species, size/age, and vigor. Streng et al. (1989) found that certain species, such as red maple (*Acer rubrum* L.), deciduous holly, and sweetgum had a higher likelihood of survival. Additionally, older seedlings demonstrated a higher survival rate; survival was lowest within the first year after germination and increased with age. Battaglia et al. (2000) also found that smaller seedlings had lower survival rates. In their study, Peterson & Bazzaz (1984) found that two-year old silver maple seedlings had greater capacity for photosynthetic recovery than two-month old seedlings. All of the older plants recovered in one week, and the only significant differences in photosynthesis after draining were found only one day after flooding for seedlings that were flooded for both 11 and 21 days. In comparison, the two-month old seedlings flooded for 11 days recovered in two weeks. A lower transpiration rate than the control during flooding, followed by a higher rate after recovery, demonstrates a decrease in water use efficiency. Two-month old seedlings flooded for 21 days had not begun to recover after two weeks and had chlorotic leaves.

NATIVE SPECIES ECOLOGY IN BLHW FORESTS

OAKS

Oak has historically been a significant component of BLHW forests in many parts of the eastern United States. However, in recent decades, oak regeneration has become a concern. Oliver et al. (2005) noted the lack of adequate red oak regeneration in the Mississippi delta floodplain to replace the present overstory composition. The height growth of red oaks was limited by overstory trees more than other species, leading to suppression and mortality and contributing to a shift in species composition. Establishment of most lowland oak forests can be traced back to some form of facilitating disturbance, such as fire or grazing. First-year oak seedlings that germinate beneath a dense canopy rely on food reserves from the acorn. Once these stores are consumed, light becomes a critical factor for seedling survival as new photosynthate production is required. Oaks are categorized as mid- to intolerant of shade, with white oaks typically more tolerant than red oaks. At the forest floor, photosynthetically active radiation is often at or below the level oaks need to maintain a positive carbon balance. If seedlings cannot replenish their carbohydrate reserves, they will die (Hodges & Gardiner 1993).

The growth strategy of oaks initially prioritizes carbon allocation to root growth rather than shoot growth. Because photosynthate is allotted to building a root system, oak seedlings may not respond to release as promptly as other genera. Additionally, intolerant competitors that designate more resources to shoot growth have a height advantage. In comparison with other species, less shoot growth occurs within the first growing season. When oak seedlings occur with more tolerant species, their comparative physiological inability to thrive in low light conditions can put them at a disadvantage and contribute to a slow juvenile growth rate (Hodges & Gardiner 1993). Collins & Battaglia (2008) found that, ten years after artificial gap creation, canopy openings did not influence initial oak establishment, as seedlings were more abundant beneath the canopy, but were important for progression into the sapling stage, as juveniles were tallest in gap centers. Additionally, they found that the initial filters on regeneration were related to seed dispersal and predation and microsite conditions. Three years in, natural oak regeneration was most abundant around the gap edges and in microsites that were undisturbed with litter cover. Swamp chestnut oak germination was impeded in pits, while cherrybark oak (*Quercus pagoda* Raf.) establishment and growth was diminished by shading and seed predation. Soil moisture can also be an influential determinant of oak regeneration; on river floodplains, density and survival may be higher on wetter micro-sites (Hodges & Gardiner 1993).

In this East Texas study, water oak (*Quercus nigra* L.), willow oak (*Quercus phellos* L.), overcup oak, and cherrybark oak are the most prominent oak species. The red oak group (water, willow, and cherrybark) tends to occur on well-drained flats and ridges (Oliver et al. 2005), while overcup oak is more tolerant of saturated soils (Kroschel et al. 2016). The seeds of these four species are dispersed primarily by water and animals, and germination occurs in the spring (Burns & Honkala 1991, Kabrick & Dey 2001).

OTHER NATIVE HARDWOODS

Other hardwood genera that are common in North American floodplain forests include *Fraxinus, Ulmus, Acer, Betula, Salix, Populus, Celtis, Platanus,* and *Liquidambar,* with species and dominance varying based upon region and successional stage (Mitsch and Gosselink 2015). From this diverse selection, green ash (*Fraxinus pennsylvanica* Marsh.), American elm (*Ulmus americana* L.), sweetgum, water hickory (*Carya aquatica* (Michx. f.) Nutt.), American hornbeam (*Carpinus caroliniana* Walt.), blackgum (*Nyssa sylvatica* Marsh.), common persimmon (*Diospyros virginiana* L.), and water-elm (*Planera aquatica* J.F.Gmel.) are the most common non-oak native tree species in this study area. Both rapid growing species, sweetgum is the most shade intolerant, and green ash is moderately tolerant to tolerant of shade. Blackgum is also shade intolerant and has a moderate growth rate. American elm and water hickory have moderate shade tolerance and moderate to slow growth rates. Persimmon, American hornbeam, and water-elm are all shade tolerant species and relatively slow growers. The light-seeded, wind and water-dispersed species include green ash, American elm, sweetgum, and water-elm. Heavy-seeded species, dispersed by birds, animals, and water, include water hickory, American hornbeam, blackgum, and persimmon.

Flood tolerance during the growing season varies among these hardwoods. Water-elm thrives in very wet soils with flooding for the longest periods; blackgum also thrives in flooded conditions and can tolerate them for longer periods. Young persimmon can be killed by prolonged flooding, but seedlings can survive adverse conditions. Water hickory tolerates prolonged spring flooding. Green ash and sweetgum are more mesic species, but can endure flooding during the growing season. American hornbeam and elm are moderately tolerant of flooding; American elm does well in moist soils but is often killed by growing season floods (Burns & Honkala 1991, Kabrick & Dey 2001).

INVASIVE SPECIES ECOLOGY IN BLHW FORESTS

SILVICS & INVASIVENESS OF CHINESE TALLOW

Invasive species are a prominent management concern in riparian ecosystems across the globe. Chinese tallow is an invasive tree species that seriously threatens BLHW forests in the southeastern United States. It originates from China/Japan, and was first brought to the east coast of North America by Benjamin Franklin in the 1770s. In the early 1900s, it was more widely introduced to the Gulf Coast in commercial plantations, but quickly escaped and spread into riparian areas. By the 1940s, it was continuing to spread further into prairies and upland ecosystems. In the period from 1991 to 2005, tallow abundance was estimated to have increased in Texas by 174%, becoming increasingly prominent in comparison to native species. It is the most abundant sapling in the Big Thicket National Preserve, and the most common species in several counties. Cold intolerance appears to be one of the factors limiting its spread, but anticipated climate change over the next century may increase its range further northwards (Bruce et al. 1997, Pile et al. 2017).

Chinese tallow flowers in late spring, from April to June. Seeds are primarily dispersed by water and birds. Birds are often an important distribution agent for invasive species, and many species consume tallow seeds. An r-strategist, it is also a heavy seeded species, which gives it a higher probability of success in late-seral communities. Tallow can reach maturation and begin flowering within a few growing seasons. These factors contribute to high propagule pressure. The rapid growth rate of Chinese tallow also helps it outcompete native species. It has been shown to exceed sugarberry (*Celtis laevigata* Willd.), which is one of the fastest-growing species in Texas. A unique feature is its ability to perform equal to or better than native species in both deep shade or forest gaps. Though it grows best in full sunlight, it can succeed in many light environments and outcompete both shade-tolerant and shade-intolerant native trees. This trait allows it to invade both early and late successional communities. Tallow seedlings have a competitive edge against oak seedlings because oaks often do not have a high enough light compensation point at the forest floor to meet their requirements (Bruce et al. 1997, Camarillo et al. 2015, Pile et al. 2017). Wall and Darwin (1999) found that tallow seemed unaffected by shading, and saplings of native overstory trees were uncommon. Chinese tallow has a relatively short lifespan, reaching ages of 100 years but often going into decline as early as 50 years (Pile et al. 2019).

Chinese tallow is able to survive under a dense overstory and in wet conditions. Seedlings gain substantial flood tolerance within the first two months of their lives. It prefers mesic environments, and invasion is more rapid in wetter sites compared to dry.

Although it does remarkably well in anaerobic conditions, it is also considered drought tolerant (Pile et al. 2017). One Texas study found that tallow survival was not affected by flood duration (Siemann & Rogers 2003). Tallow has even higher rates of mycorrhizal association in North America than in its native environment. It also conforms to the enemy release hypothesis; it experiences less herbivory in its non-native territory and thus can allocate more resources to growth and less resources to defense (Bruce et al. 1997, Pile et al. 2017). Floodplains are additionally vulnerable to tallow due to both anthropogenic and natural disturbances, such as dredging, forestry operations, nitrogen deposition, catastrophic wind events, and feral hogs (Pile et al. 2017). In an East Texas BLHW forest, Camarillo et al. (2015) found that high Chinese tallow abundance was negatively correlated with density of native species in the understory and overstory. Oaks, specifically, had low abundance in plots that had high abundance of tallow. Areas with established tallow were denser with smaller stems, indicating that they were in the stem exclusion stage, and had low stocking of oaks. In contrast, areas with less tallow presence were composed of larger native species with a prominent oak component. However, though tallow's rapid growth may give it an advantage initially after disturbance, Pile et al. (2019) found that its diameter and height growth declined substantially after eight years compared to slash pine (*Pinus elliottii* Engelm.).

EFFECTS OF INVASIVE SPECIES TREATMENTS IN RIPARIAN FORESTS

The invasion of an exotic species begins with an establishment, or “lag,” phase, where the population increases slowly. This time period can be long, and eradication of the intruder is usually only possible during this stage. The expansion phase shows exponential population growth and spread, during which the plant becomes much harder to control. Lastly, the saturation phase occurs when the spread has reached its geographic limit (Webster et al. 2006). Chinese tallow has several viable treatment methods, including herbicide, fire, mechanical, and biological control. A combination of these approaches may be the most effective at managing the population and allowing native species to establish. Herbicide application may be most effective in the late summer or early fall (Pile et al. 2017, Webster et al. 2006). Chinese tallow resprouts after being cut or burned, which can also be an important factor to consider (Bruce et al. 1997).

Treatment/removal of other invasive species in riparian systems has shown mixed results in terms of subsequent success of native species. In a Florida floodplain, Smith et al. (2016) found that repeated herbicide application on Mexican petunia (*Ruellia simplex* C. Wright) and revegetation with native species was unsuccessful, and petunia re-invaded after treatment. In another study near the Ohio River, tree seedling density steadily increased across five years after Amur honeysuckle was sprayed with herbicide. Species richness and percent cover of herbaceous species also increased, although the abundance of other invasive plants grew as well (Boyce 2015). In the Upper Mississippi River floodplain, Thomsen et al. (2012) tested how treatment of reed canary grass, along with other site preparation, would aid native tree regeneration after blowdown and salvage logging. They found that the average density of tree seedlings was approximately ten times higher in areas treated with herbicide, and establishment was low in untreated areas. Additionally, Hudson et al. (2014) evaluated recovery of a riparian forest in Georgia five years after Chinese privet (*Ligustrum sinense* Lour.) was mechanically removed. They found an increase in total plant cover, with a prevalence of early, colonizing species. Woody saplings were more prominent in removal plots, with species including boxelder (*Acer negundo* L.), green ash, and sweetgum dominating the composition.

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Chapter II: Natural Regeneration Dynamics in Relation to Abiotic and Biotic Factors in Post-Herbicide Environments

INTRODUCTION

Tree regeneration in riparian ecosystems exhibits complex dynamics due to the heterogeneity of microsites caused by unique landscape and flooding patterns (Terwei et al. 2013). In many of these forests, natural regeneration of desired species has been reduced due to a combination of issues, including land conversion (Kellison & Young 1997, Kroschel et al. 2016), altered river hydrology (Nilsson & Berggren 2000), high grading (Kellison & Young 1997), and invasive species (Predick & Turner 2008, Repka et al. 2015). The presence of desired, native advance regeneration and saplings in a forest is critical for canopy replacement in the event of a disturbance. The composition, spatial distribution, and abundance of this regeneration are affected by a wide variety of biotic (e.g., herbaceous ground cover, invasive species, overstory) and abiotic (e.g., canopy cover, microtopography, organic matter) factors (Streng et al. 1989), which play a role in maintaining species richness (Grubb 1977). Research of the initial years of a seedling's life, even the first growing season, can indicate species-specific filters on regeneration (Kroschel et al. 2016). Studying these factors in floodplain forests facilitates a deeper understanding of the intricate workings in this critical ecosystem. Additionally, factors that influence natural regeneration dynamics of native species are of great importance to management by enhancing the establishment and recruitment of desired species.

Light and flooding are known influential abiotic variables on dynamics and establishment of regeneration in floodplain forests (Battaglia et al. 2000, Battaglia & Sharitz 2006, Hall & Harcombe 1998, Kroschel et al. 2016, Küßner 2003). One East Texas study found that the density of seedlings over a year old was correlated with summer light availability (Streng et al. 1989). Flood water level was determined to be the greatest influencer on overall seedling abundance, and results indicated that density of light-seeded species was reduced by flooding, while heavy-seeded species were less impacted. Water levels in bottomland forests are partially influenced by microtopography. Battaglia et al. (2000) found in a simulated study that light and water can affect regeneration both individually and through interaction.

Regeneration adjacent to down woody debris could benefit from protection from herbivory and burial, as well as altered microsites that could provide a less stressful growing environment (Harmon et al. 1986). Cover and depth of plant litter on the forest floor also has the potential to affect the germination and establishment of seedlings. Litter can aid by enhancing the soil moisture, reducing competition, and protecting seeds from predation. However, it can also be detrimental to regeneration by reducing temperature or burying, obstructing, and shading seeds. Plant litter can increase soil nutrients by releasing them during decomposition, but compared to uplands, it may be less impactful in riparian systems because it is not present for as long (Xiong & Nilsson 1997).

Ground cover competition by tall herbaceous flora (i.e., forbs and grasses) can have an impact on tree regeneration, as they can shade out new or slow-growing germinants (Siebel & Bouwma 1998). Competition from invasive plants, in particular, can have an influence on the regeneration dynamics of native species. Alien trees affect multiple strata in forests, competing for growing space with trees in the canopy, outcompeting native juveniles when gaps open (Webster et al. 2006), and even facilitating the success of their own offspring (Terwei et al. 2013). Chinese tallow is an invasive tree species that seriously threatens bottomland hardwood (BLHW) forests in the southeastern United States. In an East Texas BLHW forest, Camarillo et al. (2015) found that high Chinese tallow abundance was negatively correlated with density of native species in the understory and overstory. Oaks, specifically, had low abundance in plots that had high abundance of tallow. Additionally, feral hogs (*Sus scrofa* L.) are a prominent biotic threat to the bottomland forests of the southern United States. As they "root" for food, they disrupt the soil and litter layers and cause mechanical damage to existing plants (Kroschel et al. 2016).

Treatment/removal of other invasive species in riparian systems has shown mixed results in terms of subsequent success of native species. In a Florida floodplain, Smith et al. (2016) found that repeated herbicide application applied to Mexican petunia (*Ruellia* *simplex* C. Wright) and revegetation with native species was unsuccessful, and petunia re-invaded after treatment. Hudson et al. (2014) evaluated recovery of a riparian forest in Georgia five years after Chinese privet (*Ligustrum sinense* Lour.) was mechanically removed. They found an increase in total plant cover, with a prevalence of early, colonizing species. Woody saplings were more prominent in removal plots.

Chinese tallow has several viable control treatment methods, including herbicide, fire, mechanical, and biological control (Pile et al. 2017a). Boggy Slough Conservation Area, the study site of this research, has been treating Chinese tallow to reduce impacts on their property via herbicide treatments since 2014. How natural regeneration occurrence and dynamics varied in the post-treatment environments remains unclear. This research assessed how regeneration dynamics were influenced by various abiotic and biotic factors in the post-treatment environment through two objectives. The first objective was to quantify current regeneration dynamics (species composition, density, and height and diameter of established seedlings and saplings) in areas previously treated with herbicide to reduce Chinese tallow occurrence and untreated, reference areas. The second objective was to evaluate the influence of abiotic and biotic factors (e.g., microtopography; down woody debris mass; overstory density; canopy cover; ground cover by leaf litter, grasses, forbs, and shrubs; density of Chinese tallow seedlings and saplings; occurrence of feral hog damage) on the occurrence of native seedlings and saplings.

METHODS

STUDY AREA

This research was conducted within the Pineywoods ecoregion of East Texas, which is divided between the Southeastern Mixed Forest Province and the Outer Coastal Plain Mixed Forest Province. The Pineywoods has a subtropical humid climate; winters are typically mild with occasional frosts, while summers are hot and humid. The average yearly temperature ranges from 64 to 67º F, with average January lows between 30 and 39º F and average July/August highs ranging from 93 to 100º F. Rainfall is abundant and uniformly dispersed throughout the year, even during the warmest parts of summer. Annual precipitation totals average from 40 to 60 inches. Certain climatic extremes, such as winter frosts and summer droughts, are important for sustaining natural flora and excluding non-adapted species. This ecoregion is characterized by a flat to gently rolling landscape that ranges from nearly zero to 500 feet above sea level (Diggs et al. 2006).

River floodplains are an important component of the Pineywoods landscape. Rivers in the area are generally slow-moving, meandering, and turbid. They regularly overflow their banks due to the abundant rainfall in the region. The alluvial soils of these floodplains are commonly classified as Inceptisol or Entisol orders, with sediment frequently deposited during flood events. With slightly higher elevations, natural levees and ridges contain sandier soils and experience shorter flooding duration, lower flooding frequency, and higher deposition rates. Corresponding tree cover includes mesic species such as water oak, white oak (*Quercus alba* L.), cherrybark oak, sweetgum, and loblolly pine (*Pinus taeda* L.). Most prominent of the bottomland ecological types, seasonally flooded river floodplains occur on the flat floodplains of major rivers, and are primarily part of a seasonally inundated hydrologic regime. Soil texture is typically loamy to clayey. More flood-tolerant deciduous hardwoods dominate these areas, including willow oak, laurel oak (*Quercus laurifolia* Michx.), overcup oak, swamp chestnut oak, American elm, red maple, sweetgum, and water hickory. Within this type, these species vary further in distribution based on subtle elevation differences. Of the floodplain oaks, overcup oak is the most flood-tolerant and typically occupies more regularly flooded areas (Diggs et al. 2006, Hodges 1997). East Texas has approximately 1.64 million acres of BLHW forests remaining along its major rivers such as the Angelina, Neches, Trinity, and Sabine (Diggs et al. 2006, Kroschel et al. 2016).

STUDY SITE

This study was conducted in a BLHW forest along the Neches River at Boggy Slough Conservation Area (BCSA), which is located approximately seven miles northeast of Apple Springs, TX. Considered the last "wild" river in Texas, the Neches River is a major waterway in East Texas, flowing approximately 416 miles from Van Zandt County to the Gulf of Mexico. The drainage of this river encompasses 10,011 mi^2, with a yearly flow of approximately six million acre-feet of water near the end of its course. There are two major reservoirs on the river; Lake Palestine is located 40 miles below the headwaters at Rhine Lake, approximately 140 miles northwest of the study site. B.A. Steinhagen Lake is roughly 86 miles southeast of the study site (TPWD 2020). The eastern boundary of BSCA is defined by the Neches River. Soils at the study site are of the Ozias-Pophers complex, which commonly occur on sites with zero to one percent slopes and frequent flooding. The Ozias series (fine, smectitic, thermic Aeric Dystraquerts) originates from clayey alluvium and is a hydric, somewhat poorly drained soil with a moderate available water capacity. The Pophers series (fine-silty, siliceous, active, acid, thermic Fluvaquentic Endoaquepts) originates from loamy alluvium and is also hydric and somewhat poorly drained, with high available water capacity. Both series are deep and slowly permeable. These series are commonly associated with hardwoods such as oaks (water, willow, overcup, etc.), elm spp., green ash, sugarberry, and sweetgum (USDA 2020).

To survey current regeneration dynamics in environmental conditions associated with previous Chinese tallow herbicide (triclopyr) treatments, sampling plots were established in areas treated in 1) 2015 and 2018 (2-TRT); 2) 2019 (1-TRT); and 3) an untreated area (REF) (Figure 2.1). A 2.5% solution of Garlon® XRT herbicide was applied via broadcast spraying in areas accessible by a tractor and with a relatively high density of small Chinese tallow stems. Hand crews used a hack-and-squirt method in areas without tractor access and variable tallow density. In these cases, a 30% solution of Garlon® XRT herbicide was used with one hack per 3 inches of tree diameter.

FIELD METHODS

PLOT LOCATION

A total of 36 plots, 12 in each treatment area, were established. Within each treatment, four transects were systematically spaced and located using Avenza Maps™ (laid out with the Draw and Measure tool). Three plots were located along each transect at distances of 164, 492, and 984 feet (50, 150, and 300 m) from the river to account for potential vegetation gradients and variation in hydroperiod and sedimentation based on fluvial landforms (Smith 1996). Distance from river was based at an approximate 90-degree angle from the Neches River. A compass and handheld GPS unit were used to navigate to each plot along a specified azimuth. Each plot center was marked with rebar, and the closest tree was tagged and distance and azimuth to the tagged tree recorded.

DATA COLLECTION

Forest structure and composition were characterized in variable-radius plots for overstory (diameter at breast height; DBH $\geq$ 4 inches) and fixed-radius (1/100th acre) plots for sapling (DBH 0.6 to 3.9 inches), seedling (DBH < 0.6 inches), and ground cover (shrub, vine, forb, grass, leaf litter, bare, woody debris) strata (Figure 2.2). Inventory data were collected in August 2020. At each plot center, a 10 BAF wedge prism was used estimate overstory basal area (ft^2/ac). Species and DBH (4.5 ft. above ground) were recorded for each tallied overstory tree.

A regeneration survey was conducted to determine density, size, and species composition of regeneration strata. At each plot center, all saplings in a 1/100th acre circular plot were tallied. Species, height, and basal diameter were recorded for saplings. Seedlings of woody tree species were tallied in the northeast and southwest quadrants of the plot (1/200th acre). All seedlings 6 inches tall (hereafter, large seedling) or greater were identified by species, and height and basal diameter were measured. Seedlings shorter than 6 inches (hereafter, small seedling) were tallied by species only due to the higher mortality rates of recently-germinated seedlings (Nemens et al. 2018). The seedling classes were distinguished separately because of their differing implications for dominance probability.

Biotic and abiotic factors were evaluated in each plot to assess their influence on the occurrence and abundance of regeneration by species and size class. Ground cover was estimated for shrubs, vines, forbs, grasses, leaf litter, bare ground, and down woody debris (DWD) using the Daubenmire class system. Six cover classes were used: 1) 0-5%, 2) 5-25%, 3) 25-50%, 4) 50-75%, 5) 75-95%, and 6) 95-100% (Daubenmire 1959). Canopy cover was considered for assigning classes to the plant growth forms, while the non-plant categories were evaluated as ground cover. This information was gathered in the northeast and southwest quadrants of the plot. The amount of feral hog damage (i.e., rutting) was also assessed with the Daubenmire classes for the entirety of the plot.

Organic matter depth was measured in N, S, E, and W directions at full radius from plot center. DWD mass (tons/ac) was estimated using a modified Brown (1974) method along the full N-S and E-W line transects of the plot; pieces were tallied if they intersected either of the transects (11.78 ft.), with size classes marked as 1-3 inches, 3-8 inches, or greater than 8 inches at the point of intersection. Canopy cover was measured with a spherical densiometer at plot center; four readings were taken facing north, east, south, and west and averaged. Microtopography was classified as either level, ridge, slope, or bowl (Almquist et al. 2002). Ridges and bowls were described as equivalent to mounds (distinctly convex) and pits (distinctly concave) as suggested by Cornett et al. (2016), respectively. Slopes represented areas that were transitional between higher and lower areas, while level areas had no noticeable variation.

STATISTICAL ANALYSES

Density per acre (basal area for overstory and stems for regeneration strata) and mean size (diameter and height) were calculated for each species by plot. The size differences among individual species were not formally tested due to uneven, and sometimes low, abundances of the species across treatments. Native species data were summarized by hypogeal and epigeal germination strategies, due to grouping of native species by heavy- and light-seeded species in previously published literature focused on bottomland regeneration dynamics (Kroschel et al. 2016, Jones et al. 1994, Streng et al. 1989). Table 2.1 displays the scientific and common names of the species codes that will be used in other tables and figures. Data were analyzed using RStudio 1.4.1106 (RStudio Team 2021) at a significance-level of $\alpha = 0.10$. This significance-level was selected due to the complexity and large number of variables in ecological research, which can cause difficulty in statistically recognizing biological significance at smaller alpha-levels. Permutation tests assessed significance in the following analyses, and the assumption of exchangeability of observations under the null hypothesis was met.

To compare differences in density (stems/acre) of species in the regeneration layers among the 2-TRT, 1-TRT, and REF areas, indicator species analysis (ISA) was conducted using the indicspecies package (De Caceres & Legendre 2009) using the multipatt function. This analysis is an effective way to examine species performances across two or more experimental units (McCune & Grace 2002), such as areas with different management histories. A separate ISA was conducted for sapling, large seedling, and small seedling size classes, identifying species that arose as indicators in certain treatment areas. The restcomb command was used to specify that the only combination of groups considered in the analysis would be 2-TRT + 1-TRT, as combinations of treated areas with the REF area were not of interest for interpretation. The number of each microtopography class in each area was examined, though not analyzed statistically. Each class was similar in number among all three areas, supporting the ability to compare them despite a lack of pre-treatment data.

Canonical Correspondence Analysis (CCA) was used to assess how measured independent abiotic and biotic variables related to regeneration densities of native species. This analysis was performed using the vegan package (Oksanen et al. 2020), using the cca function in RStudio. CCA was selected because it can be used to relate species relative abundances to quantifiable variables representing environmental gradients and can analyze variables measured on multiple scales. It was assumed that the abundance of each species follows a unimodal model, rather than a linear model, meaning that abundance is maximized at a specific set of environmental conditions. The interest of this objective was not to determine the strongest environmental gradients, but to assess the influence of the measured variables on the community structure of native regeneration, making this an appropriate technique (McCune & Grace 2002).

Sapling, large seedling, and small seedling size classes were analyzed separately. The primary goal of the CCAs was exploratory analysis; thus, many variables were considered and stepwise selection methods were employed to further examine which variables may be of interest for each size class. Variables considered included distance from river; microtopography; canopy cover; DWD mass; organic matter; percent ground cover of litter, forb, grass, shrub, and bare ground; percent hog damage, overstory basal area, and density of Chinese tallow seedlings and saplings.

The correlation matrix (Pearson's correlation coefficient; r) of all variables was estimated to identify and remove highly correlated variables. Small and large seedling size classes of Chinese tallow were combined due to an existing pairwise correlation of 0.98. The variance inflation factors (VIFs) for each variable were assessed to avoid multicollinearity; VIFs of 10 or greater signaled redundant variables in the model and were examined further. Stepwise selection methods were used to identify potentially important variables using the ordistep function. Results from the selection methods along with vector lengths in the full model biplots were also used to determine which variables to remove from the final model. In the triplots, the scaling option that optimized species was used with Hill's scaling in order to best reflect their chi-square distances and represent where species co-occur (McCune & Grace 2002). Permutation tests with 999 iterations were used to calculate p-values for the overall models, terms, and axes.

RESULTS

FOREST STRUCTURE AND COMPOSITION

OVERSTORY

Total overstory basal area and species composition was similar among treatment areas. Native hypogeal and epigeal densities, as well as those of white oak and red oak species groups, were fairly equal throughout the site (Table 2.2). Various oak species were present throughout, with overcup oak and willow oak most prevalent. Overall, the oaks were larger in diameter and likely among the older trees across the site. Among the native-epigeal species, sweetgum, green ash, and blackgum were the most abundant among all areas. Chinese tallow basal area was low (<15 ft^2/acre) among all areas and was not observed in the 2-TRT area. Chinese tallow diameters were small in the 1-TRT and REF areas, indicating that the trees likely had recently been recruited into the overstory.

REGENERATION STRATA: SAPLINGS & SEEDLINGS

Native sapling density was relatively low among all treatment areas (Table 2.3). Total sapling density was greater in the REF than in the 2-TRT (315%) and 1-TRT (222%), which was contributed to by high densities of shade-tolerant common persimmon and water-elm. Another shade-tolerant native-epigeal species, American hornbeam, was present at high densities across the site. Sapling densities of native hypogeal species were consistently lower than those of native epigeal species. Additionally, red oak saplings were more abundant than white oak saplings across all areas. Chinese tallow sapling density was higher than the density of most native species in the sapling size class, comprising 42% of the total sapling densities in 2-TRT + 1-TRT and 25% of the total sapling density across all areas. Native species showed no noticeable trends in sapling height among areas, while tallow height was lower in the 2-TRT area.

Seedling densities were higher than sapling densities. Total large seedling densities were similar among the three areas, while small seedling densities had more variation. Total seedling density was lowest in the 2-TRT area and highest in the 1-TRT area for small and large seedling size classes. Proportions of native hypogeal and epigeal species varied among treatment areas (Tables 2.4 and 2.5). Three oak species (overcup, water, and willow oaks) were observed in the seedling layers. Overcup oak was the dominant oak species in the seedling strata, and its density was greatest in the REF area. Chinese tallow seedling density was high in both treated and reference areas, comprising 42% of the total densities of both the total large and small seedling layers. The average basal diameter of tallow was greatest in the 2-TRT area. Native seedlings did not demonstrate strong trends in diameter.

INDICATOR SPECIES AMONG TREATMENT AREAS

Indicator species in the 2-TRT area emerged in the large seedling class: American elm, American hornbeam, and green ash (Table 2.6). No species arose as significant indicators for the 1-TRT area alone. Chinese tallow (large seedling) and American elm (small seedling) were indicators for the treated areas as a whole (2-TRT + 1-TRT). In the REF area, common persimmon was a sapling indicator species and the only sapling species to be an indicator overall. Overcup oak and water-elm were also found to be REF indicator species in both the small and large seedling classes.

FACTORS INFLUENCING NATIVE SPECIES ABUNDANCE

Most environmental variables were not highly correlated, with all of the Pearson's correlation coefficients below 0.7 (Table 2.7). Pairwise comparisons of variables with higher correlations, such as between percent bare ground and hog damage, percent bare ground and litter, percent forb and grass, and OM depth and percent litter, had the potential to both be included if their vectors had varying biplot scores. However, when vectors of correlated variables had similar behavior, only the strongest one was kept in the model. The sapling class had the lowest number of species in its CCA, while the small seedling class had the greatest (Table 2.9).

The small seedling model included microtopography; DWD mass; percent ground cover of litter, forb, grass, shrub, and bare ground; percent hog damage, overstory basal area, and density of Chinese tallow seedlings (Table 2.8 and Figure 2.3). The permutation test indicated that the model was significant ($p = 0.008$), and only the first two constrained axes were significant ($p = 0.001$ and $p = 0.048$, respectively). In this model, the proportion of inertia explained by the constrained axes was 0.4933, and the adjusted r-squared was 0.2203. In the small seedling class CCA, canopy cover, organic matter depth, tallow sapling density, and distance from river were excluded from the final model.

The CCA for large seedlings included microtopography; canopy cover; DWD mass; organic matter; percent ground cover of forb, grass, and shrub; percent hog damage, overstory basal area, and density of Chinese tallow saplings (Table 2.8 and Figure 2.4). The p-value of the model was significant ($p = 0.038$), and the first constrained axis was significant, while the second was not ($p = 0.048$ and $p = 0.171$, respectively). The proportion of inertia explained by the constrained axes was 0.5074, and the adjusted r-squared was 0.1849. Distance from river, percent litter, percent bare ground, and tallow seedling density were removed from the final model.

The sapling CCA, which had a smaller sample size due to less frequent occurrence of saplings, included distance from river, microtopography, canopy cover, DWD mass, percent cover of shrubs, percent hog damage, overstory basal area (Table 2.8 and Figure 2.5). The permutation test indicated that the model was significant ($p = 0.031$). The first constrained axis was significant, but the second was not ($p = 0.013$ and 0.164, respectively). The proportion of inertia explained by the constrained axes was 0.6428, and the adjusted r-squared was 0.2355. VIFs in this analysis were uncommonly high, leading to exclusion of bare ground as a variable despite its strong correlation with the first axis. Percent forb and grass, organic matter depth, and tallow seedling and sapling density were also removed from the final model.

DISCUSSION

CHINESE TALLOW ABUNDANCE AMONG TREATMENT AREAS

Chinese tallow was more abundant, emerging as an indicator species, in treated areas (2-TRT + 1-TRT) for the large seedling size class. This finding suggests that the development of newly established seedling cohorts of Chinese tallow may have been facilitated in the post-treatment environment. Plant invasions have been commonly linked to disturbance (Lozon & MacIsaac 1997), and multiple studies have demonstrated that invasion of Chinese tallow, specifically, can be enhanced by disturbance and increased light levels in the lower forest strata (Henkel et al. 2016). Pile et al. (2017b) illustrated significantly greater abundances of Chinese tallow in disturbed forests than those with less disturbance impacts. Density of tallow increased as a result of management practices that increased light levels (e.g., thinning). Increased growth rates have also been documented for Chinese tallow with increased light levels (Siemann & Rogers 2003).

Broadcast spraying and hack-n-squirt herbicide applications can influence densities of non-native plants. Strong propagule pressure from invasive species may give them an advantage over native species in acquiring newly-available resources after a disturbance such as herbicide application (Mason & French 2007, Rinella et al. 2009). Treatment-related dieback or top-kill of many larger tallow stems may increase light levels in certain areas, though not analyzed in this study. Removal of invasive European buckthorn (*Rhamnus cathartica* L.) via mowing was shown to increase light availability, which facilitated the growth and survival of its regeneration (Anfang et al. 2020). Though there is little research regarding light availability changes following herbicide treatments, the principle likely carries to other removal methods beyond mowing.

If the growth rate of tallow seedlings was increased in the post-treatment environment, it may explain why the smallest small seedling size class was not an indicator in the treated areas; much of the new regeneration may have grown into the large seedling class. Though not significant, the overall high average density of small tallow seedlings in the 1-TRT area was due to a few plots in which most of the overstory was comprised of treated tallow saplings, resulting in a thick layer of new tallow regeneration. Because tallow was only an indicator for combined treated areas (not either one individually), time since treatment was likely not a largely influential factor for this species in this study. Further, without pre-treatment data, it is unknown how the abundance of tallow seedlings in the untreated area compared to the treatment areas downriver prior to treatments (Figure 2.1). It is possible that the REF area was invaded to a lesser degree, which could have also played a role in tallow being an indicator species in the treated areas.

Though the post-treatment environment may have facilitated the growth and development of tallow seedlings, tallow saplings did not emerge as an indicator for any area. This is likely a reflection of the hack-n-squirt treatments in the treated areas. However, because a high proportion of the total sapling density was made up by tallow, some are still likely to be recruited in the future, taking the place of desired species in the canopy. It may be that without the second treatment in 2018, greater abundances of tallow saplings would have been observed in the 2-TRT area and that repeated treatments will be necessary to prevent more severe re-invasion. Long-term monitoring of regeneration at the site will be important to aid in better understanding of the impact of the post-treatment environment on both native and non-native species and if the short-term dynamics following treatments persist.

NATIVE SPECIES ABUNDANCE AMONG TREATMENT AREAS

Overcup oak was the only oak species to be an indicator and was the most abundant oak species for the seedling classes. Overcup oak was found to be an indicator for large seedlings in the REF area, which may reveal a trade-off between microenvironmental conditions favorable for tallow seedlings and overcup oak seedlings. Overcup oak seeds can germinate abundantly under a closed canopy, but ultimately perish unless released. The most flood-tolerant of the bottomland oaks (Kroschel et al. 2016), this species has intermediate tolerance to shade and competition, relying on early-season flooding to kill less flood-tolerant competition (Burns & Honkala 1990). Historically, disturbance events that increase light levels were instrumental in the genesis of many oak-dominated forests. However, when fast-growing invasive species, such as Chinese tallow, are present, it is likely that oak species will be outcompeted due to their carbon allocation patterns (Hodges & Gardiner 1993), even though light and disturbance are important factors to their recruitment. One Missouri bottomland study found that removal of the midstory and understory did not increase the density of pin oak (*Quercus palustris* Muenchh.) advanced reproduction (Motsinger et al. 2010). This finding echoes that of this study, though the removals were not strictly of invasive species.

Sapling overcup oak density was quite low in the REF area, however, indicating that overcup oak seedlings may not survive long enough or compete well enough to be positioned for canopy recruitment in many areas. This was true for the oak genus as a whole as well, ranging from approximately 25 to 142 stems/ac. among the three areas (Table 2.3), with less than 20% of plots stocked with oak indicating an uneven distribution. There are no straightforward answers regarding adequacy of oak regeneration density in BLHW forests. However, according to some guidelines in the literature, the oak sapling density at BSCA may not be adequate to facilitate the persistence of an oak-dominated stand. Recommendations range from approximately 40 to 400 stems/ac. in the advance regeneration layer, with greater numbers accounting for more-realistic higher mortality rates (Lockhart et al. 2000, Oliver et al. 2005). Clatterbuck and Meadows (1992) suggested 150 free-to-grow oaks/ac. before a regeneration harvest, while Sander (1979) recommended 400 stems/ac. of at least 4.5 feet or greater in height before overstory removal. Additionally, an even distribution of oaks throughout the stand may also be important; guidelines have advised that at least 60% of plots should be stocked to better assure success of regeneration; this was not met at this site (Stanturf & Meadows 1994). Many of the recommendations in the literature were created primarily for red oaks, but can be applied to a red oak white-oak mix as well (Meadows & Stanturf 1997). In light of this, “adequacy” of desired oak saplings was greatest in the REF area, but still may not be sufficient to ensure dominance of this genus in the future depending on mortality rates and recruitment opportunities.

Both treated areas and the REF area had the same number of native species to arise as indicators across all size classes. Shade-tolerant species, common persimmon and water-elm were also found to be indicator species in the REF area. The presence of persimmon as an indicator species is likely a reflection of the high abundance of shade-tolerant species in the sapling layer. A slow growing species, it can persist in the understory for many years (Burns & Honkala 1990). Water-elm as a seedling indicator likely follows similar reasoning. A lack of major disturbance in this area, failing to open large swaths of the canopy, will lead to persistence of shade-tolerant species and likely their eventual replacing of early-colonizers in the canopy (Hardin & Wistendahl 1989, Hodges 1997, King & Antrobus 2005).

Green ash and American hornbeam, which are also considered tolerant of shade, were seedling indictors in the 2-TRT area, along with intermediate American elm. This may indicate that the aftermath of either multiple treatments or longer time, since the initial treatment results in higher occurrences of these species as seedlings. It is likely that the treatments were not a severe enough disturbance for more intolerant, early-successional species to arise as indicators in their aftermath. Hudson et al. (2014) found that American hornbeam was an indicator species in "desired future condition" plots five years after the removal of Chinese privet, but it was not an indicator species in the removal plots. Their results conflict with the results of this study, although the size classes were not specified in the privet study. A study in central New York wetlands observed that hornbeam was present in the understory (DBH <4 in.) in gaps created by elm mortality for multiple vegetation types (Huenneke 1983), indicating that it will opportunistically regenerate when light becomes available. Groninger et al. (2004) observed that herbicide treatments of surrounding flora increased the size of planted green ash seedlings. This study analyzed density, rather than size, but this may provide evidence that green ash responds positively to such operations. More shade-tolerant species, including green ash and American elm, are predicted to become more dominant components of previously intolerant-composed floodplain forests (Kroschel et al. 2016).

These light-seeded species can volunteer in planted bottomland restoration projects (McLane et al. 2012), so their appearance with available resources in this instance seems reasonable.

FACTORS INFLUENCING NATIVE SPECIES ABUNDANCE

Variables with high correlations to the CCA axes were considered the strongest components of those environmental gradients. The gradients that these axes appear to represent are light/canopy, ground cover, and edaphic environments. Multiple variables measured in this study could be interpreted as proxies for light availability (e.g., canopy cover, basal area, percent grass and forb, tallow density), ground cover and potentially edaphic conditions (e.g., percent grass and forb, litter/OM depth, bare ground, hog damage, tallow seedling density), or both as these are inherently related. This made clear interpretations of each axis difficult, especially as some variables were highly correlated with both.

In the small seedling ordination, bare ground was negatively correlated with the first axis, while litter and tallow seedling density were positively correlated. Overstory BA and forb cover were both positively correlated with the second axis. The first axis seems to represent a light and ground cover gradient, while the second may represent effects of microtopography and edaphic conditions. For the large seedling ordination, overstory BA and forb cover were positively correlated with the first axis, and shrub cover was negatively correlated; this may have represented a gradient related to overstory tree density. Canopy cover and forb cover were positively correlated with the second axis, and tallow sapling density and organic matter were negatively correlated with it. Siebel and Bouma (1998) found that the tall herbaceous layer was reduced by shading, which allowed woody seedlings to establish. It appears in these results that forb and grass cover did not prevent species from germinating (reflected by the small seedling CCA). Few large seedlings were in their greatest abundance in areas with high forb cover, however, perhaps also demonstrating that forb cover can hinder seedling establishment. Canopy opening is necessary for recruitment of mature trees, but advance reproduction beforehand is important, as herbaceous species can quickly establish in high light environments and outcompete new germinants (Siebel & Bouma 1998).

Shrub and litter cover and distance from river were the strongest variables loading on the first axis in the sapling ordination. Litter had a negative correlation with it and the other two positive. This axis likely represented correlation of species with microtopography and hydrology. Hog damage was positively correlated with the second axis, while basal area was negatively correlated. Shrub cover, primarily swamp privet (*Forestiera acuminata* (Michx.) Poir.) and possumhaw, was anecdotally observed in slightly wetter areas and its vector was often associated with bowls in the triplots. Its association with distance from river may represent the influence of landscape position and hydrology on sapling abundances.

Light has commonly been referenced as one of the most influential factors determining regeneration dynamics in floodplain forests (Battaglia et al. 2000, Hall & Harcombe 1998, Kroschel et al. 2016, Streng et al. 1989, Terwei et al. 2013). The lack of influence of canopy cover, however, was surprising in the small seedling CCA. The short-lived nature of most small seedlings (Jones et al. 1994) may contribute to this variable, specifically, not being a requirement for their existence in certain locations. Soil or edaphic conditions may be a more important factor for germination and initial survival, while light becomes more critical for persistence. The plots measured in this study, by random chance, also had a relatively low variation of values for canopy cover, which may have also prevented stronger trends from being observed. Streng et al. (1989) had the same issue with light intensity when analyzing seedling survival. Despite this, they found that light still influenced the density of seedlings over a year old, indicating that subtle differences in light increased in importance as seedlings aged. Similarly, in this study, canopy cover emerged as a strong vector in the large seedling CCA. Many large seedling species were more abundant in areas that did not have high canopy cover.

Canopy cover may have had a stronger influence in the sapling CCA if the variation had not been further reduced by the small number of plots. McCarthy & Evans (2000) found that overhead canopy cover did not influence the distribution of overcup oak saplings, but light did as a result of south-facing canopy gaps. The inclusion of canopy gaps may be an important aspect of the light gradient(s) that was not captured in this study. Though the canopy vector was short and the biplot scores relatively small, abundances of four of the ten sapling species were maximized in areas with lower canopy cover and only one species (water-elm) was maximized in areas with high canopy cover. This may reflect the necessity of some light for prolonged survival and/or recruitment of saplings.

Flooding depth and duration greatly influence vegetation patterns in floodplains (Battaglia et al. 2000, Hall & Harcombe 1998, Kroschel et al. 2016, Smith 1996, Streng et al. 1989). In this study, microtopography was the primary variable assessed that may capture these important variables. The classification method used did not necessarily correspond to fluvial landform positions (i.e., “level” plots were classified in both drier and wetter sites), but still captured areas where water may be more likely to collect for longer periods. In these results, microtopography appeared to have the most pronounced effect on saplings. Some species, such as overcup oak, water-elm, and American hornbeam demonstrated fairly consistent preference for the same positions across size classes, while most of the species varied some.

Hog damage did not appear to be a detriment to regeneration, as there were a number of species with high abundance in areas with higher percentage of rooting in all size classes. It may be that saplings were less affected by rutting due to their size. The observed damage in the plots may have been old, which allowed new seedling layers to germinate in the time since. Jones et al. (1994) found that plots with approximately 25% rooting did not appear to influence seedling density, mortality, or recruitment. Another study in lower Michigan found that hog-disturbed plots tended to have lower tree regeneration, though not significant. However, hog disturbance did not appear to influence overall composition and structure of woody plants over two years (Gray et al. 2020). In contrast, another study found that heavy-seeded species increased in abundance in hog-excluded areas (Siemann et al. 2009).

Overcup oak, in contrast to its relatives, was most abundant in bowls or flats across all size classes, which is consistent with its silvical characteristics. It is one of the more flood-tolerant bottomland species, commonly found in poorly drained areas (Burns & Honkala 1990, McCarthy & Evans 2000). It was commonly associated with water-elm, another flood-tolerant species. Overcup oak–water hickory forests often follow water-elm in the successional path of poorly-drained sites (Hodges 1997), so it is reasonable that these two species had greater abundances in similar microtopographical positions. Litter depth has been observed to have a negative effect on establishment of overcup oak (McCarthy and Evans 2000), which was not strongly supported by these results; large seedling abundance was maximized in areas with medium organic matter depth.

Red oak species in the seedling classes did not conform to consistent patterns. As a small seedling, water oak abundance was maximized with high levels of tallow seedling density, percent litter, and overstory BA. As a large seedling, however, water oak was more often found in areas with high canopy, shrub, and grass cover. Small willow oak abundances were not strongly preferential of conditions, but as a large seedling it was more abundant with low canopy and shrub cover. Willow oaks were most abundant on ridges or slopes as seedlings and slopes as saplings. Water oak and cherrybark oak were most abundant on ridges in the sapling class. Oliver et al. (2005) stated that species in the red oak group are commonly found on ridges and well-drained areas with the most productive soils. These sapling species also tended to occur in (relatively) medium to low canopy cover, reflecting the requirement for some light to develop beyond the seedling stage.

CONCLUSIONS

This research did not evaluate the efficacy of the herbicide treatments, due to a lack of pre-treatment data, but examined dynamics in their aftermath. Composition in bottomland forests at BSCA may be experiencing an overall shift from more shade-intolerant species, such as oak, sweetgum, or hickory, towards more shade tolerant species. Overcup oak, the most common oak in the seedling layer, was higher in abundance in the REF area. The seedling strata had the most indicator species in the post-treatment environment, which appeared to facilitate new cohorts of Chinese tallow, perhaps at the expense of overcup oak. Green ash and American elm, non-oak desirable native species, were also more abundant in treated areas. Results represent one point in time, and future research will be important to determine if/how the observed dynamics persist over longer amounts of time. Regeneration of most species varied in their preferences of environmental conditions across size classes. Light, ground cover, and microtopography variables represent important resources that can influence the establishment success and distribution of BLHW species.

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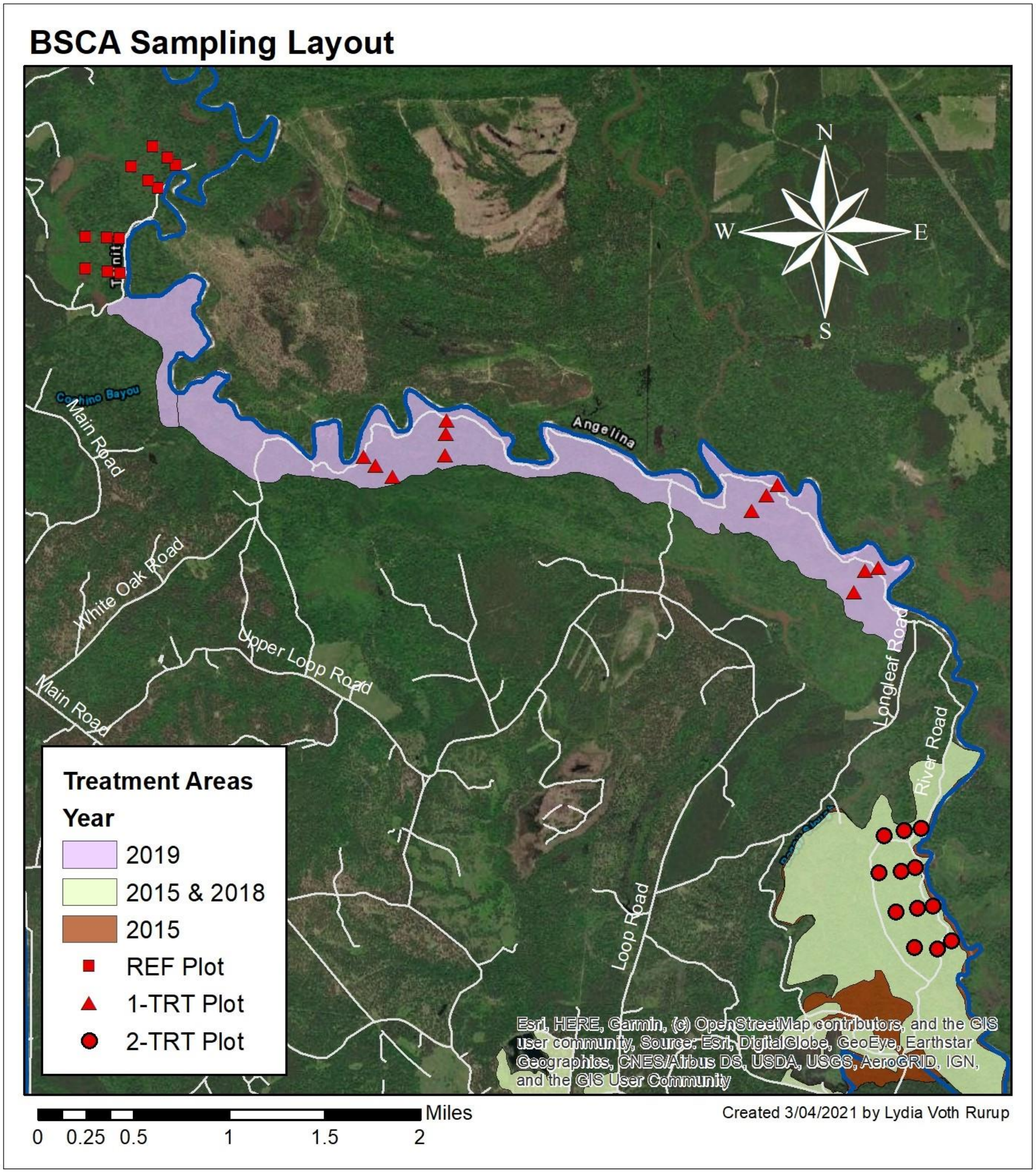

Figure 2.1 Sampling layout of 36 plots at Boggy Slough Conservation Area (BSCA) within two areas previously treated once (1-TRT; 2019) and twice (2-TRT; 2015 & 2018) with herbicide for Chinese tallow and one untreated area (REF). Plots were systematically spaced 164, 492, and 984 feet (50, 150, and 300 m) from the Neches River in East Texas.

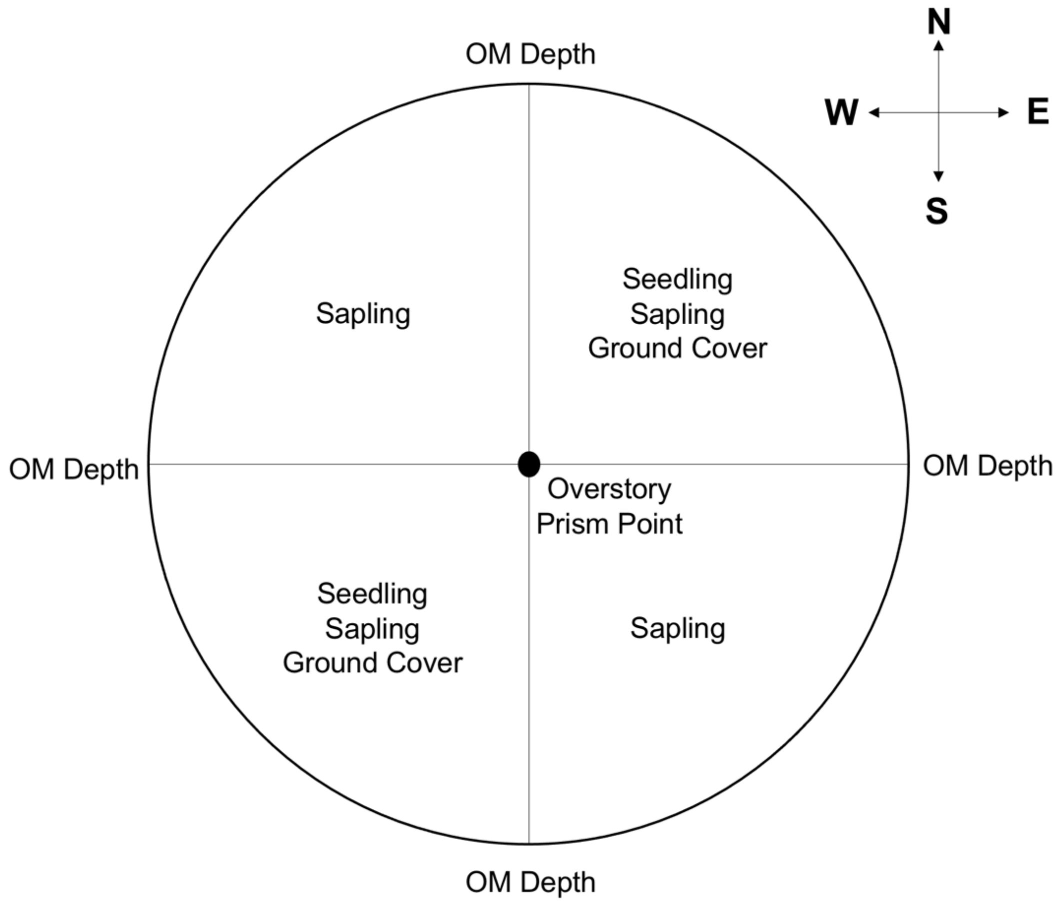

Figure 2.2 Locations of overstory, regeneration, ground cover, and organic matter (OM) measurements around the 1/100th acre circular plot.

Table 2.1 Scientific and common names that correspond to the species codes for tree species recorded at Boggy Slough Conservation Area.

Species code	Scientific name	Common name
CAAQ	*Carya aquatica*	water hickory
CACA	*Carpinus caroliniana*	American hornbeam
CRMA	*Crataegus marshallii*	parsley hawthorn
DIVI	*Diospyros virginiana*	American persimmon
FRPE	*Fraxinus pennsylvanica*	green ash
GLAQ	*Gleditsia aquatica*	water locust
ILOP	*Ilex opaca*	American holly
LIST	*Liquidambar styraciflua*	sweetgum
NYSY	*Nyssa sylvatica*	blackgum
PITA	*Pinus taeda*	loblolly pine
PLAQ	*Planera aquatica*	water-elm
QULY	*Quercus lyrata*	overcup oak
QUMI	*Quercus michauxii*	swamp chestnut oak
QUNI	*Quercus nigra*	water oak
QUPA	*Quercus pagoda*	cherrybark oak
QUPH	*Quercus phellos*	willow oak
QUSI	*Quercus similis*	bottomland post oak
TRSE	*Triadica sebifera*	Chinese tallow
ULAL	*Ulmus alata*	winged elm
ULAM	*Ulmus americana*	American elm

Table 2.2 Mean overstory (DBH ≥ 4 inches) basal area (feet²/acre) and DBH (inches) with standard errors in parentheses (where applicable, > 1 plot) for bottomland hardwood stands treated once (1-TRT; 2019) and twice (2-TRT; 2015 & 2018) with herbicide for Chinese tallow and untreated (REF) at Boggy Slough Conservation Area. Species codes listed in Table 2.1.

	Basal area (ft.²/ac.)			DBH (in.)		
Species	**2-TRT**	**1-TRT**	**REF**	**2-TRT**	**1-TRT**	**REF**
Native-Hypogeal						
White oaks	10.8 (3.6)	21.7 (5.6)	15.0 (4.7)			
QULY	8.3 (3.7)	19.2 (6.1)	15.0 (4.7)	21.6 (3.9)	25.9 (1.4)	23.4 (4.0)
QUMI	0.0 (0.0)	0.8 (0.8)	0.0 (0.0)	.	22.8	.
QUSI	2.5 (1.8)	1.7 (1.7)	0.0 (0.0)	16.0 (3.0)	17.8	.
Red oaks	24.2 (6.1)	18.3 (6.0)	15.0 (5.1)			
QUNI	9.2 (3.4)	2.5 (1.3)	1.7 (1.7)	26.7 (3.6)	35.3 (7.0)	24.1
QUPA	5.8 (2.9)	0.8 (0.8)	0.0 (0.0)	35.7 (5.9)	37.5	.
QUPH	9.2 (3.4)	15.0 (5.8)	13.3 (4.0)	26.2 (2.3)	30.4 (2.9)	27.8 (3.8)
Non-oak						
CAAQ	1.7 (1.1)	0.0 (0.0)	1.7 (1.7)	11.8 (4.2)	.	21.05
Total	36.7 (5.9)	40.0 (7.3)	31.7 (5.3)			
Native-Epigeal						
CACA	2.5 (1.3)	0.8 (0.8)	0.8 (0.8)	5.7 (1.1)	5.5	4.35
DIVI	0.0 (0.0)	1.7 (1.7)	2.5 (1.8)	.	11.2	7.6 (3.5)
FRPE	5.8 (2.6)	4.2 (2.3)	16.7 (6.1)	17.5 (2.9)	19.3 (1.3)	17.1 (2.2)
ILOP	4.2 (2.9)	0.0 (0.0)	0.0 (0.0)	10.9 (2.7)	.	.
LIST	25.0 (6.7)	25.0 (13.2)	3.3 (1.9)	23.6 (2.5)	22.4 (2.3)	10.8 (3.2)
NYSY	10.8 (3.6)	5.0 (2.9)	0.0 (0.0)	20.4 (1.6)	21.7 (1.1)	.
PITA	1.7 (1.7)	4.2 (2.3)	1.7 (1.7)	36.5	36.2 (2.4)	38.3
PLAQ	0.8 (0.8)	3.3 (1.9)	5.0 (2.3)	16.1	12.8 (2.8)	13.4 (2.6)
Other [a]	0.8 (0.8)	0.0 (0.0)	0.8 (0.8)	.	8.3	4.5
Total	51.7 (8.5)	44.1 (13.3)	30.8 (8.1)			

(Continued)

Invasive-Epigeal						
TRSE	0.0 (0.0)	14.2 (8.9)	5.0 (2.9)	.	6.5 (0.84)	8.1 (0.8)
Total	88.3 (8.2)	98.3 (12.1)	67.5 (10.1)			

[a] Native-epigeal species present in less than 3% of plots (CRMA and ULAM) were classified as “other.”

Table 2.3 Mean sapling (DBH 0.6 to 3.9 inches) density (stems/acre) and height (feet) with standard errors in parentheses (where applicable, > 1 plot) for bottomland hardwood stands treated once (1-TRT; 2019) and twice (2-TRT; 2015 & 2018) with herbicide for Chinese tallow and untreated (REF) at Boggy Slough Conservation Area. Species codes listed in Table 2.1.

Species	**Density (stems/ac.)**			**Height (ft.)**		
	2-TRT	**1-TRT**	**REF**	**2-TRT**	**1-TRT**	**REF**
Native-Hypogeal						
White oaks						
QULY	0.0 (0.0)	0.0 (0.0)	58.3 (39.8)	.	.	17.0 (3.0)
Red oaks	25.0 (17.9)	66.7 (66.7)	83.3 (74.7)			
QUNI	0.0 (0.0)	58.3 (58.3)	0.0 (0.0)	.	11.6	.
QUPA	16.7 (16.7)	0.0 (0.0)	0.0 (0.0)	22.1	.	.
QUPH	8.3 (8.3)	8.3 (8.3)	83.3 (74.7)	7.2	7.8	14.1 (3.6)
Non-oak						
CAAQ	0.0 (0.0)	8.3 (8.3)	0.0 (0.0)	.	.	.
Total	25.0 (17.4)	74.9 (66.4)	141.7 (82.1)			
Native-Epigeal						
CACA	100.0 (56.4)	216.7 (199.2)	16.7 (16.7)	14.0 (2.8)	13.4 (2.2)	11
DIVI	0.0 (0.0)	0.0 (0.0)	108.3 (52.9)	.	.	12.6 (2.0)
FRPE	0.0 (0.0)	0.0 (0.0)	16.7 (11.2)	.	.	13.3 (4.8)
LIST	41.7 (22.9)	8.3 (8.3)	41.7 (33.6)	15.1 (8.9)	8.3	26.0 (8.8)
PLAQ	0.0 (0.0)	16.7 (16.7)	608.3 (599.3)	.	10.6	14.1 (0.6)
Other [a]	0.0 (0.0)	8.3 (8.3)	8.3 (8.3)	.	13	16.4
Total	141.7 (57.0)	250.0 (196.8)	800.0 (585.3)			
Invasive-Epigeal						
TRSE	175.0 (131.5)	172.7 (103.3)	133.3 (66.7)	8.0 (2.7)	21.7 (6.6)	19.1 (2.2)
Total	341.7 (170.8)	483.3 (275.2)	1075.0 (603.5)			

[a] Native-epigeal species present in less than 3% of plots (CRMA and ULAL) were classified as “other.”

Table 2.4 Mean large seedling (DBH < 0.6 inches, height ≥ 6 inches) density (stems/acre) and basal diameter (inches) with standard errors in parentheses (where applicable, > 1 plot) for bottomland hardwood stands treated once (1-TRT; 2019) and twice (2-TRT; 2015 & 2018) with herbicide for Chinese tallow and untreated (REF) at Boggy Slough Conservation Area. Species codes listed in Table 2.1.

Species	**Density (stems/ac.)**			**Basal Diameter (in.)**		
	2-TRT	**1-TRT**	**REF**	**2-TRT**	**1-TRT**	**REF**
Native-Hypogeal						
White oaks						
QULY	450.0 (181.1)	300.0 (219.5)	2483.3 (641.7)	0.15 (0.02)	0.12 (0.02)	0.12 (0.01)
Red oaks	133.3 (116.3)	183.3 (133.6)	50.0 (35.9)			
QUNI	16.7 (16.7)	0.0 (0.0)	0.0 (0.0)	0.14	.	.
QUPH	116.7 (116.7)	183.3 (133.6)	50.0 (35.9)	0.25	0.14 (0.06)	0.08 (0.02)
Non-oak						
CAAQ	33.3 (22.5)	16.7 (16.7)	50.0 (35.9)	0.18 (0.09)	0.13	0.16 (0.03)
Total	616.7 (186.6)	500.0 (244.3)	2583.3 (673.5)			
Native-Epigeal						
CACA	350.0 (216.2)	33.3 (33.3)	16.7 (16.7)	0.24 (0.06)	0.05	0.07
DIVI	66.7 (51.2)	200.0 (132.6)	283.3 (124.2)	0.16 (0.02)	0.28 (0.08)	0.23 (0.05)
FRPE	116.7 (52.0)	16.7 (16.7)	16.7 (16.7)	0.21 (0.06)	0.13	0.45
LIST	66.7 (37.6)	83.3 (57.5)	0.0 (0.0)	0.30 (0.09)	0.15 (0.06)	.
NYSY	116.7 (86.9)	0.0 (0.0)	0.0 (0.0)	0.18 (0.00)	.	.
PLAQ	0.0 (0.0)	33.3 (22.5)	400.0 (293.4)	.	0.14 (0.05)	0.15 (0.05)
ULAM	616.7 (251.6)	83.3 (57.5)	16.7 (16.7)	0.19 (0.03)	0.09 (0.00)	.
Total	1333.3 (450.1)	450.0 (194.0)	733.3 (294.7)			
Invasive-Epigeal						
TRSE	1033.3 (263.8)	3066.7 (2634.3)	366.7 (172.0)	0.29 (0.08)	0.13 (0.02)	0.10 (0.02)

(Continued)

Total	2966.7 (733.5)	4000.0 (2998.4)	3683.3 (852.6)

Table 2.5 Mean small seedling ((DBH < 0.6 inches, height < 6 inches) density (stems/acre) and standard error (in parentheses) of bottomland hardwood stands treated once (1-TRT; 2019) and twice (2-TRT; 2015 & 2018) with herbicide for Chinese tallow and untreated (REF) at Boggy Slough Conservation Area. Species codes listed in Table 2.1.

Species		**2-TRT**	**1-TRT**	**REF**
Native-Hypogeal				
White oaks				
QULY		1066.7 (468.6)	283.3 (133.6)	7883.3 (2685.7)
Red oaks		16.7 (16.7)	33.3 (33.3)	66.7 (51.2)
QUNI		0.0 (0.0)	0.0 (0.0)	16.7 (16.7)
QUPH		16.7 (16.7)	33.3 (33.3)	50.0 (50.0)
Non-oak				
CAAQ		33.3 (22.5)	0.0 (0.0)	50.0 (35.9)
	Total	1116.7 (462.8)	316.6 (140.3)	8000.0 (2739.7)
Native-Epigeal				
CACA		166.7 (54.1)	183.3 (86.9)	100.0 (67.4)
DIVI		83.3 (45.8)	483.3 (282.3)	633.3 (260.3)
FRPE		50.0 (35.9)	7133.3 (4609.1)	2766.7 (1029.8)
LIST		100.0 (52.2)	150.0 (60.9)	50.0 (35.9)
PLAQ		0.0 (0.0)	50.0 (35.9)	1683.3 (738.3)
ULAM		866.7 (376.1)	500.0 (265.7)	16.7 (16.7)
Other [a]		66.7 (51.2)	33.3 (22.5)	16.7 (16.7)
	Total	1333.3 (399.5)	8533.7 (4574.3)	5266.6 (1226.8)
Invasive-Epigeal				
TRSE		566.7 (98.0)	15183 (11565.3)	1716.7 (414.5)
Total		3016.7 (517.3)	24033.3 (14996.5)	14983.3 (3569.8)

[a] Native-epigeal species present in less than 3% of plots (GLAQ, ILOP, NYSY, and PITA) were classified as "other."

Table 2.6 Significant p-values ($p < 0.10$) of small seedling (DBH < 0.6 inches, height < 6 inches), large seedling (DBH < 0.6 inches, height ≥ 6 inches), and sapling (DBH 0.6 to 3.9 inches) indicator species in areas treated once (1-TRT; 2019) and twice (2-TRT; 2015 & 2018) with herbicide for Chinese tallow and untreated (REF) at Boggy Slough Conservation Area. Species codes listed in Table 2.1.

Strata/Indicator Species	**2-TRT**	**2-TRT + 1-TRT**	**REF**
Small Seedlings			
ULAM	-	0.0016	-
PLAQ	-	-	0.0001
QULY	-	-	0.0155
Large Seedlings			
ULAM	0.0105	-	-
CACA	0.0383	-	-
FRPE	0.0739	-	-
TRSE	-	0.0442	-
QULY	-	-	0.0027
PLAQ	-	-	0.0158
Saplings			
DIVI	-	-	0.0251

Table 2.7 Pearson's correlation coefficients (r) between 12 parametric abiotic and biotic variables, collected in 36 plots that were considered in the canonical correspondence analyses. Means, medians, and ranges of these variables are listed at the bottom of the table.

Variable	Canopy cover	DWD mass	OM depth	Shrub	Forb	Grass	Litter	Bare	Hog	Basal area	TRSE seedling	TRSE sapling
Canopy cover	1.000											
DWD mass	-0.532	1.000										
OM depth	0.135	0.002	1.000									
Shrub	0.080	0.250	-0.048	1.000								
Forb	-0.243	-0.106	-0.318	-0.215	1.000							
Grass	-0.477	0.332	-0.362	-0.079	0.576	1.000						
Litter	0.239	-0.185	0.629	-0.199	-0.409	-0.445	1.000					
Bare	0.154	-0.029	-0.437	0.278	-0.106	-0.016	-0.591	1.000				
Hog	0.084	-0.164	-0.271	0.059	0.126	0.040	-0.307	0.547	1.000			
Basal area	0.130	-0.226	-0.109	-0.220	0.056	-0.229	-0.140	0.304	0.156	1.000		
TRSE seedling	-0.105	-0.067	-0.117	-0.079	-0.080	-0.096	-0.130	-0.154	-0.091	0.111	1.000	
TRSE sapling	-0.405	0.190	0.158	-0.110	-0.184	-0.144	0.079	-0.029	-0.163	-0.073	-0.072	1.000
Unit	%	tons/ac.	in.	%	%	%	%	%	%	ft^2/ac.	stems/ac.	stems/ac.
Mean	94.5	34.0	0.3	10.7	11.1	9.8	43.1	30.2	16.5	84.7	7305.6	158.3
Median	97.9	6.4	0.2	2.5	2.5	2.5	31.9	26.3	2.5	85.0	1800.0	0.0

(Continued)

Range												
low	67.6	0.4	0.0	2.5	2.5	2.5	2.5	2.5	2.5	20.0	0.0	0.0
high	99.8	369.9	1.1	61.3	85.0	62.5	97.5	91.3	97.5	170.0	173600.0	1600.0

Table 2.8 A comparison of biplot scores (numerical variables) and centroids for factor constraints (microtopography classes; Mt.) among three size classes of natural regeneration for the first two CCA ordination axes. Eigenvalues and proportion of variance explained by the constrained axes are included above the variables.

	SMALL SEEDLING		**LARGE SEEDLING**		**SAPLING**	
	AXIS		AXIS		AXIS	
	1	*2*	*1*	*2*	*1*	*2*
Eigenvalue	0.631	0.441	0.486	0.370	0.933	0.705
Proportion explained	0.433	0.303	0.313	0.238	0.312	0.236
Variables [a]						
Canopy cover	-	-	-0.377	0.453	0.267	0.076
DWD mass	-0.417	-0.282	-0.035	-0.267	-0.310	0.027
OM depth	-	-	**-0.061**	**-0.576**	-	-
Shrub	**-0.526**	**-0.485**	**-0.787**	**0.139**	0.908	0.350
Forb	**-0.360**	**0.483**	**0.547**	**0.637**	-	-
Grass	-0.419	-0.198	-0.072	0.186	-	-
Litter	**0.547**	**0.026**	-	-	-0.600	-0.421
Bare	-0.561	0.166	-	-	-	-
Hog	-0.370	0.399	0.282	-0.062	**-0.549**	**0.504**
Basal area	0.533	0.587	0.527	0.066	-0.119	-0.406
TRSE seedling	**0.526**	**0.075**	-	-	-	-
TRSE sapling	-	-	0.380	-0.663	-	-
Distance from river	-	-	-	-	**0.724**	**0.321**
Mt_Level	0.013	0.012	0.074	-0.064	0.113	0.111
Mt_Slope	0.462	0.665	-0.504	-0.700	-0.414	-2.116
Mt_Ridge	-0.399	2.146	0.412	-0.828	-1.369	1.107
Mt_Bowl	-0.026	-0.336	-0.198	0.356	0.375	-0.448

[a] Bolded terms were significant at $\alpha = 0.10$. The order of the terms as they are added sequentially into the model will influence their significances.

Table 2.9 A comparison of species biplot scores between three size classes of native natural regeneration for the first two CCA ordination axes. Species codes listed in Table 2.1.

	SMALL SEEDLING		LARGE SEEDLING		SAPLING	
	AXIS		AXIS		AXIS	
	1	*2*	*1*	*2*	*1*	*2*
Species						
CAAQ	-0.700	0.075	-0.115	-0.543	-	-
CACA	-0.457	1.354	1.265	-1.407	-1.294	0.592
DIVI	0.082	0.418	0.017	-0.416	-0.233	-1.737
FRPE	0.926	0.041	0.670	-0.378	0.660	-0.117
GLAQ	0.364	-0.123	-	-	-	-
ILOP	-0.226	2.047	-	-	-	-
LIST	-0.388	1.116	0.946	-0.176	-0.247	-0.989
NYSY	-1.960	3.359	1.832	1.789	-	-
PITA	-0.427	0.000	-	-	-	-
PLAQ	-0.733	-1.093	-2.477	0.558	0.946	0.357
QULY	-0.653	-0.328	-0.349	0.001	0.594	-0.605
QUNI	1.305	0.613	-0.685	1.575	-1.360	0.824
QUPA	-	-	-	-	-1.219	1.439
QUPH	-0.338	0.536	0.700	-1.523	-0.535	-1.673
ULAL	-	-	-	-	-0.587	-1.100
ULAM	-1.136	2.064	1.407	1.370	-	-

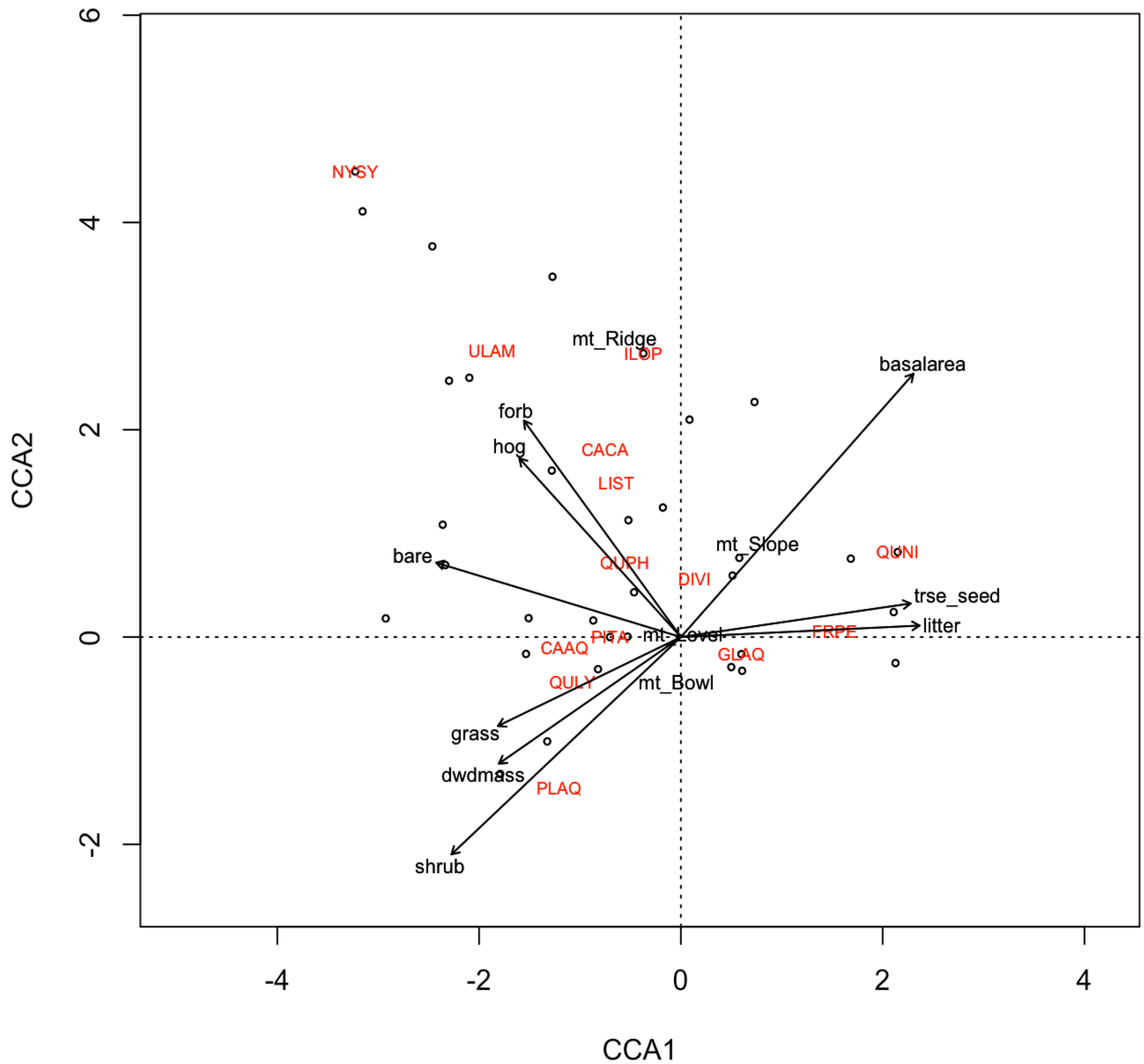

Figure 2.3 CCA ordination of 34 plots and 14 native small seedling species found throughout the bottomland hardwood forest at BSCA with 10 environmental variables. The ordination is scaled by species with Hill's scaling and the linear combination scores for sites are displayed. Environmental variables deemed to be the most explanatory were included in the triplot. Species codes listed in Table 2.1.

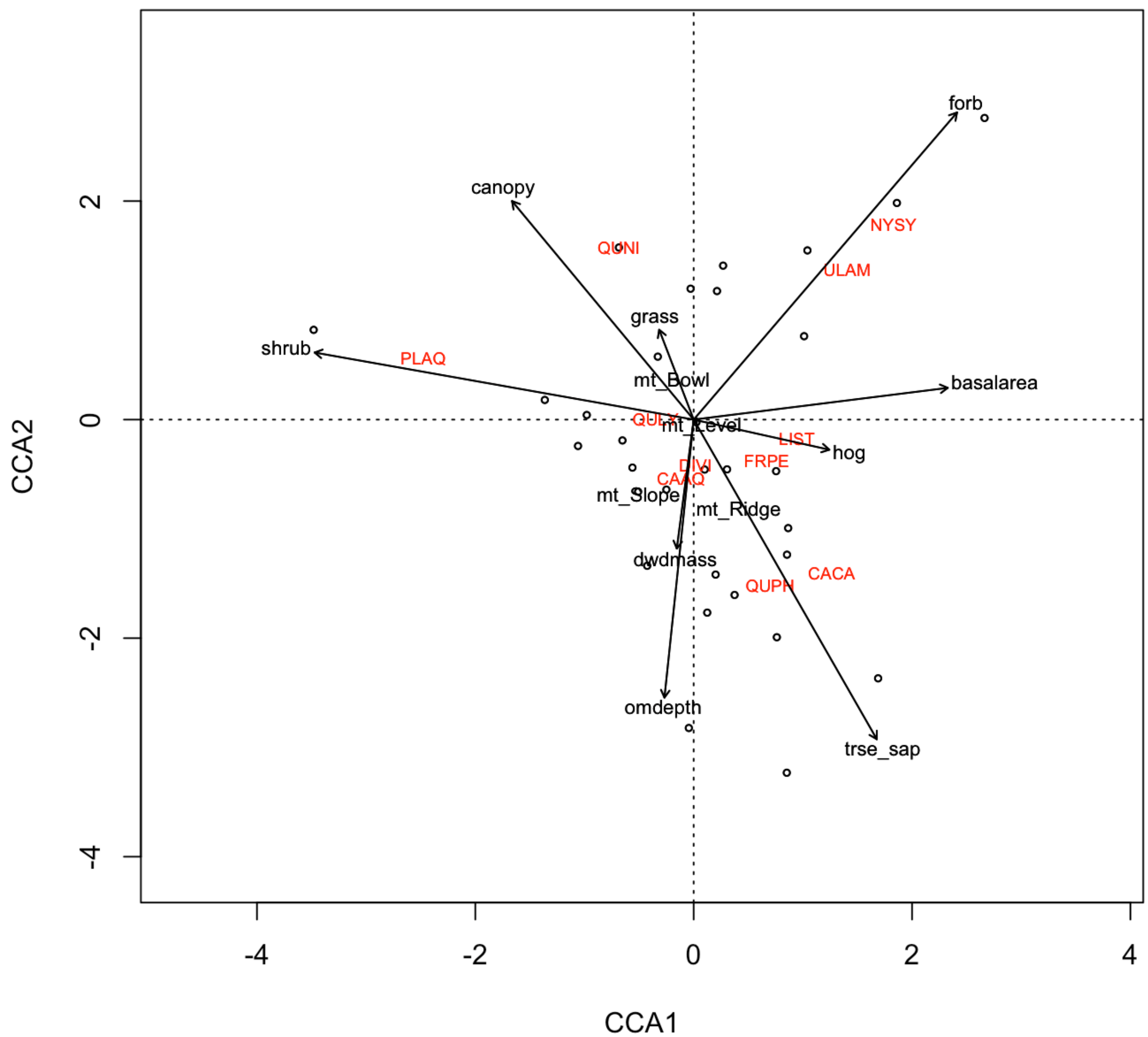

Figure 2.4 CCA ordination of 31 plots and 11 native large seedling species found throughout the bottomland hardwood forest at BSCA with 10 environmental variables. The ordination is scaled by species with Hill's scaling and the linear combination scores for sites are displayed. Environmental variables deemed to be the most explanatory were included in the triplot. Species codes listed in Table 2.1.

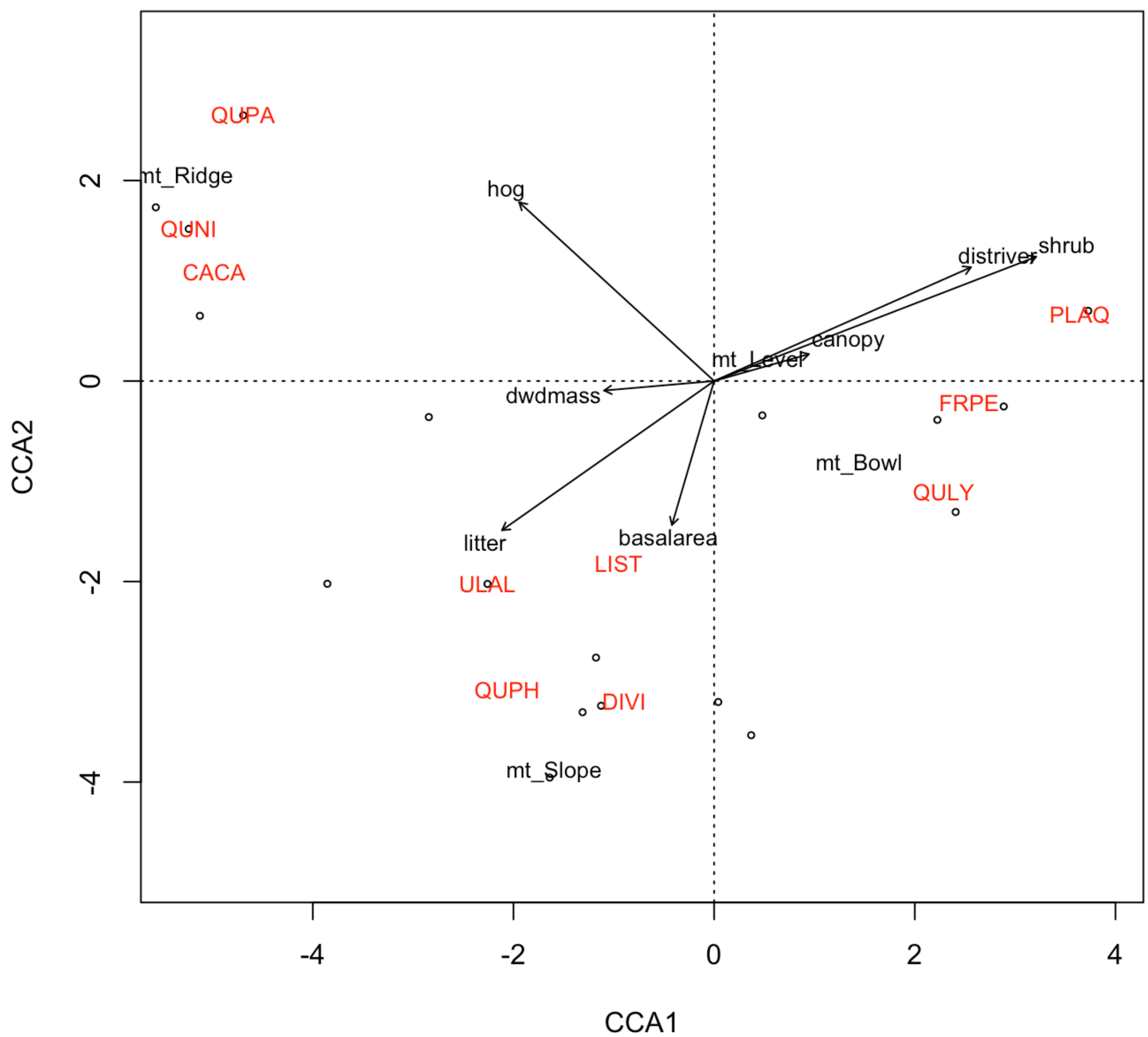

Figure 2.5 CCA ordination of 19 plots and 10 native sapling species found throughout the bottomland hardwood forest at BSCA with 8 environmental variables. The ordination is scaled by species with Hill's scaling and the linear combination scores for sites are displayed. Environmental variables deemed to be the most explanatory were included in the triplot. Species codes listed in Table 2.1.

Chapter III: One-Year Survival of Native and Invasive Natural Regeneration in a Seasonally-Flooded Bottomland Hardwood Forest

INTRODUCTION

The regeneration of desired, native species is critical to the future of healthy forest ecosystems. Trees reproduce via seed through the stages of seed production, dispersal, germination, establishment, and survival (Kroschel et al. 2016). The latter two stages ultimately determine future species composition and successional patterns. Not only is understanding where various species occur important, but how their individual characteristics and microsite conditions affect their ability to establish and persist in those locations is critical to determining future conditions of forests. In riparian forests, seedlings must endure stress from flooding as well as other forest stressors such as low light availability, competition, and herbivory (Jones & Sharitz 1998). Increased knowledge of which factors impact the survival of species groups of interest can facilitate better management of these valuable ecosystems.

Environmental factors (e.g. microtopography/flooding, light, substrate) play a key role in the survival of regeneration as they influence stress levels on regeneration (Battaglia et al. 2000, Hall & Harcombe 2001, Kroschel et al. 2016, Küßner 2003, Lin et al. 2004, Streng et al. 1989). Tolerance to these stressors can influence compatibility of species with different microsites (Jones et al. 1989). Hall & Harcombe (2001) found that, over multiple years, flooding intensity was correlated with mortality of saplings of various size classes. In an East Texas study, the variation in flooding response in light-seeded species was attributed to microsite-level differences in elevation. Elm and maple, for example, had higher mortality levels during high flooding in lower areas, while American hophornbeam and deciduous holly in the highest microsites were not affected by flooding. Alongside the lack of oxygen, damage to seedlings by floating debris may also be a leading cause of regeneration mortality (Streng et al. 1989). Light levels can also impact survival. As juveniles mature into the sapling stage, the importance of light may grow while the impact of flooding diminishes. Canopy gaps increase light penetration to the forest floor, releasing less shade-tolerant species (Kroschel et al. 2016). The presence of advance regeneration under a closed canopy is often a significant aspect of overstory replacement and recruitment. In riparian forests, seed deposition and germination may be high in adverse microsites when water levels are low, but initial densities in such areas may be reduced due to the strong influence of flooding (Huenneke & Sharitz, 1986, Jones & Sharitz 1998).

Individual seedling survival can also be influenced by the silvics of a species, size/age, and vigor (Battaglia et al. 2000, Marquis 1982, Jones & Sharitz 1998, Peterson & Bazzaz 1984, Streng et al. 1989). Species have varying tolerances to stressful conditions such as anaerobiosis or low light levels, which can impact their inherent chances of survival in floodplain systems. For instance, Streng et al. (1989) found that certain species, such as red maple, deciduous holly, and sweetgum had a higher likelihood of survival. Age also played a role in mortality patterns; older seedlings demonstrated a higher survival rate, while survival was lowest within the first year after germination and increased with age. Battaglia et al. (2000) also found that smaller seedlings had lower survival rates. In their study, Peterson & Bazzaz (1984) found that two-year old silver maple seedlings had greater capacity for photosynthetic recovery after flooding than two-month old seedlings. All of the older plants recovered in one week, and any significant differences in photosynthesis after draining were found only one day after flooding for seedlings that were flooded for both 11 and 21 days. In comparison, the two-month old seedlings flooded for 11 days recovered in two weeks. Two-month old seedlings flooded for 21 days had not begun to recover after two weeks and had chlorotic leaves.

Some native species groups may struggle to survive in the current conditions of many bottomland hardwood (BLHW) forests. Hypogeal genera such as oaks, for example, can germinate beneath a dense canopy and rely on food reserves from the acorn. Once these stores are consumed, however, light becomes a critical factor for seedling survival as new photosynthate production is required. At the forest floor, photosynthetically active radiation is often at or below the level oaks need to maintain a positive carbon balance. If seedlings cannot replenish their carbohydrate reserves, they will die (Hodges & Gardiner 1993). Epigeal species, which utilize different establishment strategies, may have higher likelihoods of survival in different conditions than hypogeal species.

In contrast, invasive species that tolerate broad conditions may have an advantage in survival. Chinese tallow is able to survive under a dense overstory and in wet conditions. Seedlings gain substantial flood tolerance within the first two months of their lives. It prefers mesic environments, and invasion may be more rapid in wetter sites compared to dry. Although it does very well in anaerobic conditions, it is considered drought tolerant. One Texas study found that tallow survival was not affected by flood duration (Camarillo et al. 2015, Pile et al. 2017).

Knowledge of regeneration survival patterns is vital in understanding which species are projected to be recruited and occupy the forest canopy in the future. While long-term research is important, annual survival studies can also help inform management decisions and provide a basis for future studies. Through this research, the status of native and invasive species in an East Texas BLHW forest was assessed through tracking survival over a one-year period in order to better understand where and how regeneration may best succeed. The specific objectives of this study were to: 1) compare one-year survival (2020-2021) rates among native hypogeal, native epigeal, and invasive Chinese tallow species groups for selected small and large seedlings and saplings and 2) determine the influence of silvical characteristics, stem size (height and diameter) and vigor, canopy cover, and micro-location (microtopography, proximity to DWD) on one-year survival of native hypogeal, native epigeal, and invasive Chinese tallow seedlings and saplings.

METHODS

STUDY AREA AND SITE

This research was conducted within the Pineywoods ecoregion of East Texas, which is divided between the Southeastern Mixed Forest Province and the Outer Coastal Plain Mixed Forest Province. The Pineywoods has a subtropical humid climate, with average yearly temperature ranging from 64 to 67º F. January lows average between 30 and 39º F, while average July/August highs range from 93 to 100º F. Rainfall is abundant and uniformly dispersed throughout the year, precipitation averaging from 40 to 60 inches annually. Certain climatic extremes, such as winter frosts and summer droughts, are important for sustaining natural flora and excluding non-adapted species (Diggs et al. 2006).

River floodplains comprise a significant portion of the Pineywoods landscape. At slightly higher elevations in bottomland forests, natural levees and ridges contain sandier soils and experience shorter flooding duration, lower flooding frequency, and higher deposition rates. Corresponding tree cover includes mesic species such as water oak, white oak, cherrybark oak, sweetgum, and loblolly pine. Most prominent of the bottomland ecological types, seasonally flooded river floodplains occur on the flat floodplains of major rivers, and are primarily part of a seasonally inundated hydrologic regime. Soil texture is typically loamy to clayey. Species including willow oak, laurel oak, overcup oak, swamp chestnut oak, American elm, red maple, sweetgum, and water hickory dominate these areas. Within this type, these species vary further in distribution based on subtle elevation differences (Diggs et al. 2006, Hodges 1997).

The study took place in a seasonally-flooded BLHW forest at Boggy Slough Conservation Area (BSCA), which is located approximately seven miles northeast of Apple Springs, TX, along the Neches River. The Neches River is a major waterway in East Texas, flowing approximately 416 miles from Van Zandt County to the Gulf of Mexico (TPWD 2020). Soils at the study site are of the Ozias-Pophers complex, which commonly occur on sites with zero to one percent slopes and frequent flooding. The Ozias series (fine, smectitic, thermic Aeric Dystraquerts) originates from clayey alluvium and is a hydric, somewhat poorly drained soil with a moderate available water capacity. The Pophers series (fine-silty, siliceous, active, acid, thermic Fluvaquentic Endoaquepts) originates from loamy alluvium and is also hydric and somewhat poorly drained, with high available water capacity. Both series are deep and slowly permeable (USDA 2020).

DATA COLLECTION

To evaluate annual survival as influenced by abiotic and biotic factors, seedling and sapling survival was monitored at the beginning and end of one year. Natural regeneration was selected and marked in the August 2020 in a total of 36 plots. Plots were established along systematically spaced transects, angled approximately perpendicular to the Neches River. Transects were located using Avenza Maps™ and laid out with the Draw and Measure tool. Three plots were located along each transect at distances of 164, 492, and 984 feet (50, 150, and 300 m) from the river to account for variation in fluvial landforms and corresponding vegetation. A compass and handheld GPS unit were used to navigate to each plot along a specified azimuth. Each plot center was marked with rebar and the closest tree was tagged and distance and azimuth to tagged tree recorded. Sampling occurred in two areas treated with herbicide for Chinese tallow (2-TRT, treated in 2015 and 2018; 1-TRT, treated in 2019) and one untreated area (REF), with four transects in each.

Three hardwood species groups were targeted in the field: desired native hardwood (oaks), other native hardwood (persimmon, green ash, water hickory, sweetgum, etc.), and invasive (Chinese tallow) species. Within each species group, two stems were targeted within three different size classes: small seedlings (0.5 to 2 ft. tall), large seedlings (greater than 2 ft. tall, but less than 0.6 inches at DBH), and saplings (0.6 to 3.9 inches DBH). If available, a total of 18 stems were selected and tagged per plot. Small seedlings were marked with pin flags, while larger regeneration with some branching was marked using zip-ties. Stems were mapped by recording azimuth and distance to the selected stem from the plot center. All stems were located within a 1/4th acre (58.9 ft. radius) from plot center. Stems were systematically located by starting in a northern direction and moving counterclockwise throughout the area, with effort made to distribute selection throughout the plot. The location of stems ceased when all possible regeneration meeting species group and size class requirements had been located and marked.

Species was recorded, and height and basal diameter were measured for each selected stem. Additionally, percent normal foliage and percent dieback were rated in ten percent increments to classify vigor for each stem. Normal foliage was defined as the percent of existing leaves without damage (herbivory, mechanical damage, etc.), discoloration (chlorosis or necrosis), or wilt/leaf curl. Dieback was described as the percent of branch/stem mortality throughout the overall specimen. A vigor class of 1, 2, or 3 was assigned based on these two percentages (Table 3.1) (Booker 2008, Dunn 1999). The microtopographical position and proximity to down woody debris were noted for each individually tagged stem. Microtopography was classified as level (no noticeable elevation shift), slope (transition from higher to lower area), raised (distinctly convex, including treefall mounds), or depression (distinctly concave, including treefall pits) (Almquist et al. 2002, Collins & Battaglia 2008). Proximity to DWD was categorized as "yes" or "no" and determined as "yes" if a stem was within one foot of a medium to large log or within a large pile of debris made up smaller woody pieces.

In August 2021, all plots were revisited, and tagged regeneration stems were relocated. Survival was determined based on the presence of a living root system; either a remaining stem with living cambium or a top-killed stem with basal/root sprouts was required. Nonliving or missing stems with a relocated, tagged marker were classified as dead. Missing stems without a tagged marker were handled on a case-by-case basis based upon the certainty of the location and presence of candidates of the same species and size in the vicinity. Height, basal diameter, and vigor were remeasured on the surviving stems.

Height was taken to the terminal point of stems, regardless of dieback. Two densiometer readings, facing north and south, were taken at each stem location and averaged to quantify canopy cover. Changes to the physical environment, such as persistence of DWD, were also noted. Flood data (gauge height) from the closest Neches River monitoring station in Diboll, TX (approximately 13 miles south of the study site) was accessed using the USGS National Water Dashboard to characterize general local flooding patterns between August 2020 and August 2021.

STATISTICAL ANALYSES

For analysis, water hickory was combined with the oak group so that native hypogeal and epigeal species could be evaluated separately. Data were analyzed at a significance-level of $\alpha = 0.10$ using SAS® Studio (SAS Institute Inc., Cary, North Carolina). The proportion of regeneration survival was compared among species groups (native hypogeal, native epigeal, invasive). The PROC FREQ function was used to run chi-square tests for association. Assumptions for this test, including usage of count data, mutually exclusive levels, independent groups, and nominal categories, were met. An initial test was run with all three species groups first to test the null hypothesis that all proportions were equal. Further, three pairwise chi-square tests were done to provide multiple comparisons between the three species groups. After total survival was tested, further comparisons were made among species groups within each size class (small seedling, large seedling, sapling) using the same process.

Separate logistic regression models were estimated to predict one-year survival for each species group using the PROC LOGISTIC function. The dependent variable, one-year survival, was binary, with a value of 1 assigned for living rootstock and 0 for dead stems (Küßner 2003). In all three logistic analyses, stem height and basal diameter, vigor, canopy cover, microtopography, and proximity to DWD were considered as potential predictor variables. Due to an inherently high Pearson's correlation ($r = 0.9$) between stem height and basal diameter, separate full and reduced models were run with those variables to avoid multicollinearity. Other assumptions for logistic regression (independence, large sample size, lack of extreme outliers, and linear relationship between independent variables and log odds) were met.

For each species group analysis, a total of six predictor variables were used to estimate the full model (Tables 3.3 and 3.4). For the categorical variables, reference categories of medium vigor, level ground, and lack of proximity to DWD were set to determine how the other categories affected the odds of survival. Treatment area was included in the Chinese tallow analysis, as potential residual effects of the herbicide that targeted tallow regeneration was of interest. Species flood tolerance and shade tolerance were also variables included in the hypogeal and epigeal analyses, respectively (Table 3.2). The silvical characteristic used for each group was chosen based on interest and one that would have a variety of species for each class; only one was chosen per group in order to reduce the number of variables included. Due to quasi-complete separation of data points in the native-epigeal model, the "raised" microtopography category was grouped with “level” and the “intermediate” shade tolerance class was grouped with “intolerant” for the epigeal analysis.

Parameter values of $p \leq 0.10$ were used to determine which predictor variables were carried into a reduced model, as they signified parameter estimates significantly different from zero. The Akaike Information Criterion (AIC) and -2 Log Likelihood (-2LL) were used to identify the best-fit models. Delta (Δ) AIC (maximum AIC – minimum AIC) values greater than 2.0 indicated whether reduced models had more support than full models. Odds ratios represented the change in the odds of survival due to a one-unit increase in a predictor variable (parametric variables) or change from a reference level (categorical variables) and were examined to identify the specific influence of each predictor variable.

RESULTS

SURVIVAL RATES AMONG SPECIES GROUPS

The null hypothesis that the proportions of alive and dead stems were equal among species groups was rejected, and it was concluded that the survival rate was not equal among native-hypogeal, native-epigeal, and Chinese tallow species groups (Chi-Square = 80.835; $p < 0.0001$). Native species groups did not significantly differ in survival rate (Chi-Square = 0.181; $p = 0.671$). Chinese tallow survival rate, however, was significantly lower than both native-hypogeal (Chi-Square = 50.745; $p < 0.0001$) and native-epigeal (Chi-Square = 58.545; $p < 0.0001$) species groups. Survival rates showed the same trend among groups in all three different size classes with sapling-sized stems demonstrating the greatest survival (Figure 3.1). When comparing among size classes, overall chi-square tests were significant at $p < 0.0001$.

CHINESE TALLOW SURVIVAL

Chinese tallow had a survival rate of 41% across all size classes (Table 3.2). All full and reduced height and diameter models were statistically significant. Convergence criteria (GCONV 10^{-8}) were satisfied in all models. For full height and diameter models, height and diameter (Tables 3.5 and 3.6), microtopography ($p = 0.008$ & 0.016), and treatment area ($p = 0.000$ & 0.001) demonstrated significant influence on tallow survival. Though overall vigor significance was greater than 0.100 ($p = 0.129$ & 0.204), a level of vigor was significant ($p = 0.054$), and thus it was included in the reduced height model. Proximity to DWD ($p = 0.154$ & 0.113) and canopy cover ($p = 0.309$ & 0.683) did not have estimates significantly different from zero in the full height and diameter models, respectively, and thus were not included in the reduced models.

ΔAIC was under 2.0 for both height and diameter, indicating that reduced models did not have substantially more support than full models. The reduced height model had the lowest AIC statistic and was significant at $p < 0.0001$, confirming that the model with the predictors was significantly better than the model with the intercept alone. Odds ratios indicated that as height growth increased by 0.5 feet, the odds of tallow regeneration survival increased by 11%. The odds of survival increased by 125% for high-vigor tallow regeneration compared to medium. Tallow stems on raised positions had 173% higher odds of survival, while those on slopes had 69% lower odds of survival compared to those on level ground. Additionally, stems in the 1-TRT and 2-TRT areas had 183% and 678% higher odds of survival, respectively, than those in the REF area (Table 3.5).

NATIVE-HYPOGEAL SURVIVAL

The native-hypogeal species group was comprised of primarily oaks and water hickory, and had an overall survival rate of 81%. All full and reduced models were significant. Convergence criteria (GCONV 10^{-8}) were satisfied in all models. Height and diameter (Tables 3.7 and 3.8), vigor ($p = 0.060$ & 0.066), and proximity to DWD ($p = 0.005$ & 0.005) had estimates that were significantly different from zero in the full height and diameter models, respectively, and thus were included in reduced models. Microtopography ($p = 0.329$ & 0.312), canopy cover ($p = 0.137$ & 0.142), and species flood tolerance ($p = 0.315$ & 0.222) were not determined to have a significant influence on hypogeal survival in either height or diameter full models, respectively, and thus were not included in reduced models.

The reduced height model was determined to be the best-fit model for hypogeal species based on the lowest AIC statistic, which was supported by a ΔAIC of 3.17. The model was significant at $p < 0.0001$. The odds ratios indicated that as height growth increased by 0.5 feet, the odds of native-hypogeal regeneration survival increased by 46%. Odds of survival for stems of both high and low vigor classes decreased by 72%

and 79% respectively. Stems growing close to DWD had 77% lower odds of survival (Table 3.7).

NATIVE-EPIGEAL SURVIVAL

The native-epigeal species group was dominated by persimmon, hornbeam, green ash, sweetgum, and water-elm, along with several other species. The overall survival rate was 82% (Table 3.2). All full and reduced models were significant. Convergence criteria (GCONV 10^{-8}) were satisfied in all models. Height and diameter (Tables 3.9 and 3.10) demonstrated significant influence on epigeal survival in the full height and diameter models. Canopy cover was included in both the reduced height and diameter models as it was statistically significant ($p = 0.096$) in the reduced diameter model and close ($p = 0.120$) in the reduced height model. Vigor ($p = 0.521$ & 0.584), proximity to DWD ($p = 0.890$ & 0.838), microtopography ($p = 0.628$ & 0.652), and species shade tolerance ($p = 0.607$ & 0.681) were not significant in the full height and diameter models, respectively, and were not carried into reduced models.

Based on ΔAIC, both reduced height and diameter models were better supported than the full models. The reduced height model was also determined to be the best-fit model for native-epigeal species according to the AIC criterion, and was significant at $p < 0.0001$. As height growth increased by 0.5 feet, odds of survival increased by 38%. Odds of survival decreased by 4% with every 5% increase in canopy cover (Table 3.9).

However, canopy cover was only below p = 0.10 in the reduced diameter model, though the odds ratio was the same in both (Table 3.10).

DISCUSSION

Stem size (or, by proxy, age) is a well-supported influencer of tree regeneration survival, and a general trend of a decrease in mortality over time has been demonstrated (Jones et al. 1994, Jones & Sharitz 1998, Kunstler et al. 2009, Marquis 1982, Peterson & Bazzaz 1984, Streng et al. 1989). Results in this study agreed with this trend, as the odds of survival for all species groups increased with stem height and diameter (Figure 3.2). Larger/older regeneration likely has a greater capacity for photosynthetic recovery after flooding, which is a key facet of flood tolerance (Anella & Whitlow 2000, Peterson & Bazzaz 1984). In floodplains, foliage on taller regeneration may not have to endure potentially fatal submersion for as long (Hosner 1958, Jones et al. 1989, Lin et al. 2004), and has a lower risk of burial by sediment or debris. Taller stems often also have better access to sunlight (Kunstler 2009). Jones & Sharitz (1998) found that height of bottomland seedlings positively influenced short-term survival, and long-term survival in some species.

Flooding has a number of physiological effects on trees that influence their survival. Soil inundation and anaerobiosis can lead to growth suppression, impede photosynthesis, and cause inefficient respiration rates that cannot produce adequate cellular energy. Insufficient energy, among other impacts of flooding, can lead to reduced uptake of mineral nutrients. Furthermore, flooding can elicit plant hormone imbalances and production of phytotoxic compounds that can cause injury. These effects cause stress that, if conditions persist, can lead to mortality of tree regeneration (Kozlowski 2002).

Along with size, vigor, microtopography, and treatment area influenced tallow survival. While low and medium vigor seedlings did not have significantly different odds of survival, regeneration with high vigor were more likely to survive. These stems were likely better equipped to handle stressors. Additionally, the odds of survival were greater in both treated areas, indicating that any residual impacts of herbicide were not a leading cause of mortality. Slopes negatively influenced survival compared to level ground, though depressions did not. Chinese tallow has demonstrated considerable flood tolerance in other studies, even compared to native bottomland species, possibly due to a high leaf-to-stem mass ratio (Butterfield et al. 2004, Connor 1994, Jones & Sharitz 1990). Thus, the impact of slope is unlikely to be related to waterlogging. However, reduced stress from flooding on higher microsites (which had better survival than level areas) may have aided tallow survival in the face of compound stress from extreme climatic events between August 2020 and August 2021.

Survival rate for Chinese tallow was significantly lower than that of native species. Abnormally high amounts of snow and ice occurred in February 2021 which may have been a contributing factor to the unexpectedly low tallow survival rate. In the Lufkin area, record low temperatures of 4º F, approximately 4.5 inches of snow/sleet, and

0.38 inches of ice were recorded between February 15-17 (National Weather Service 2021). Frost and cold temperatures have been demonstrated to be an inhibitor of Chinese tallow range expansion (Pile et al. 2017, Renne et al. 2001), so it is very likely that an unseasonable winter event could negatively affect this species' survival rate. Additionally, spring rain and flooding levels were above average in May, June, and July of 2021 (Table 3.11). Variation in year-to-year climatic and hydrological events may influence survival among cohorts more strongly than intrinsic site conditions (Jones & Sharitz 1998). However, because survival data were not collected and analyzed in relation to climate or flooding data, this conclusion cannot be fully supported in this study.

Native regeneration survival was fairly high, despite prolonged flooding during the early-to-mid growing season. Vigor and down woody debris affected the odds of native-hypogeal survival, as well as stem size. The specific influence of vigor was partially unexpected, as both high and low vigor appeared to decrease survival odds. It is reasonable that the regeneration that appeared to be struggling and weakened in 2020 would succumb to additional stressors more easily (Jones & Sharitz 1998). An explanation for the high vigor result cannot be given as simply; however, a higher percentage of the high-vigor stems were 0.1 inches or less in diameter compared to the other two classes and thus may have been more susceptible to other mortality agents. Flooding likely caused the debris to shift and move, damaging or crushing the nearby seedlings (Huenneke & Sharitz 1986, Streng et al. 1989, Yanosky 1982). The reason that only hypogeal species appeared to suffer in proximity to DWD is unclear, but a possible reason could be the higher sample size of hypogeal species growing near it compared to the other groups. The large, water-dispersed seeds of oak and hickory may be more likely to be trapped by DWD and germinate. The potential benefits of DWD offered in many forested wetlands, such as nurse logs, increased elevation above water, and varied micro-conditions, may be less reliable in floodplains due to the seasonal movement from floodwaters. Additionally, more stable woody sources such as stumps or tree bases may provide similar benefits without these risks (Sharitz 1996).

Consistent with the results of this study, Jones et al. (1994) found that shade and flood tolerance rankings did not appear to influence within-year survival patterns of regeneration. This was attributed to potential change in this characteristic over time for some species and the more likely effects over many years rather than in one growing season. Canopy cover had a slight negative impact on native-epigeal survival. Though not quite significant, canopy cover had the same odds ratio for the native-hypogeal group. Light availability, affected by overstory canopy, influences the ability of stems to produce enough photosynthate to maintain a positive carbon balance (Kunstler 2009). Measuring canopy via the densiometer at elbow-height did not always accurately capture overhead canopy at the top of taller saplings; this limitation may have prevented the full impact of the canopy from being captured. Streng et al. (1989) found that light intensity only had a minor influence on survivorship of seedlings, which was thought to be due to the low range captured in the study. Lin et al. (2004) observed higher probability of mortality in extreme low-light for shade intolerant species, with chances of survival quickly improving as light level increased. Gap dynamics may also be an important aspect of the light environment that impacts survival not quantified in this study (Hall & Harcombe 2001).

Longer-term data may be necessary to observe more patterns in light, microtopography, and/or differences between native species on reproduction survival and probability of dominance. For instance, parameter estimates for depressions were negative for these groups, though insignificant (although, the hypogeal group estimate was significant in the full diameter model). Over multiple years, it may be that the effect of flooding stress will become more pronounced on some species, more clearly identifying unfavorable locations. This would also allow for the relationships between microsite conditions and broad climatic patterns.

CONCLUSIONS

Though the microtopography classes attempted to capture local microsite variation, flooding & sedimentation variation between broad fluvial landforms not captured in this study (e.g., levees/ridges, low flats, old oxbow lakes) may also have influenced survival odds. However, this one-year data did demonstrate several potential filters that may impact successful establishment and survival of native and non-native regeneration. Individual stem characteristics (i.e., size and vigor) showed the most consistent influence on survival among groups, with microsite variables (i.e., canopy, microtopography, debris) showing more minor, less-consistent impact; longer-term data may be necessary to see more pronounced effects of this factors. The poor survival rate of Chinese tallow from 2020 to 2021 may offer native species a temporary reprieve from invasive competition. However, though tallow density may have been reduced by mortality, high rates of tallow seed production (Pile et al. 2017) may lead to germination and growth of abundant new cohorts in the newly-available growing space and, in turn, overwhelm native species. This data should serve as the beginning of a long-term effort to understand dominance probability and, thus, future canopy recruitment in modern-day BLHW forests.

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Table 3.1 Vigor classification system for regeneration stems based on percent normal foliage and percent dieback. Modified from Dunn 1999.

Vigor Class	**Definition**	**Criteria**
1	High vigor	Normal foliage ≥ 80% and dieback ≤ 10%
2	Medium vigor	Does not meet class 1 or 3 criteria
3	Low vigor	Normal foliage ≤ 20% and/or dieback ≥ 50%

Table 3.2 Bottomland hardwood species within three species groups tagged and monitored for survival over one year at Boggy Slough Conservation Area.

Group/species	**Tagged (# stems)**	**Alive [a] (# rootstocks)**	**Dead (# rootstocks)**	**# Missing**	**Flood/ shade tolerance [b]**
Invasive					
Chinese tallow	199	70	101	28	
Native - Hypogeal					Flood tol.
Overcup oak	72	31	18	23	Moderate
Willow oak	51	46	3	2	Weak
Water oak	11	9	2	0	Weak
Cherrybark oak	8	6	2	0	Weak
Bottomland post oak	1	1	0	0	Weak
Water hickory	33	23	3	7	Moderate
Total	176	116	28	32	
Native - Epigeal					Shade tol.
Common persimmon	50	34	12	4	Very tolerant
American hornbeam	24	20	3	1	Very tolerant
American holly	7	5	0	2	Very tolerant
Green ash	24	20	3	1	Tolerant
Water-elm	22	16	2	4	Tolerant
American elm	6	5	0	1	Intermediate
Parsley hawthorn	1	1	0	0	Intermediate
Sweetgum	32	24	6	2	Intolerant
Water locust	2	1	1	0	Intolerant
Total	168	126	27	19	

[a] Alive trees were classified by either a living stem or living rootstock with at least one sprout. Missing trees were not able to be relocated confidently, either due to incorrect location notes or the presence of multiple candidates in the vicinity but a missing marker, and were excluded from analysis.
[b] Silvical characteristic of interest for the analysis of each species group is reported (Burns & Honkala 1990, McKnight et al. 1980).

Table 3.3 Descriptions of variables included in logistic regression models predicting one-year survival of bottomland hardwood regeneration.

Variable	Description
Survival	Dependent binary variable; living (1) or dead (0) rootstock
Height	Total height (nearest 0.5 ft.) of stem
Diameter	Basal diameter (nearest 0.1 in.) of stem
Vigor	Vigor class of 1 (high vigor), 2 (medium vigor), or 3 (low vigor)
Canopy cover	Percent canopy cover at stem location (nearest 5%)
Microtopography	M.T.; level ground (no topography variation), slope (transition from higher to lower area), depression (distinctly concave), or raised (distinctly convex)
Proximity to DWD	Yes (stem growing within a foot of down woody debris) or no (stem not near down woody debris)
Treatment area	Areas where Chinese tallow was treated with herbicide; tallow stems tagged in REF (untreated), 2-TRT (area treated in both 2015 and 2018), or 1-TRT (area treated once in 2019)
Flood tolerance [a]	Flood tolerance (weak or moderate) of native-hypogeal species tagged
Shade tolerance [b]	Shade tolerance (very tolerant, tolerant, intolerant) of native-epigeal species tagged

[a] McKnight et al. 1980
[b] Burns & Honkala 1990

Table 3.4 Mean and range of continuous variables and number of stems of categorical variables used in predicting one-year survival of regeneration in three species groups in a bottomland hardwood forest. Variable information included in Table 3.3.

	Invasive - Chinese tallow			**Native - hypogeal**			**Native - epigeal**		
Variable	**Mean**	**Range**	**# of stems**	**Mean**	**Range**	**# of stems**	**Mean**	**Range**	**# of stems**
Height (ft.)	6.6	0.5 – 30.7	-	5.8	0.5 – 30.8	-	6.4	0.6 – 34.7	-
Basal Diameter (in.)	1.0	0.03 – 5.80	-	0.8	0.04 – 4.90	-	0.9	0.08 – 5.00	-
Canopy cover (%)	88.8	24.4 – 99.8	-	91.3	53.0 – 99.8	-	90.9	45.2 – 99.8	-
Vigor									
1	-	-	49	-	-	58	-	-	85
2	-	-	100	-	-	69	-	-	51
3	-	-	22	-	-	17	-	-	18
Microtopography									
Level	-	-	71	-	-	70	-	-	75
Slope	-	-	25	-	-	30	-	-	33
Raised	-	-	40	-	-	21	-	-	13
Depression	-	-	35	-	-	23	-	-	33
Proximity to DWD									
Yes	-	-	13	-	-	26	-	-	15
No	-	-	158	-	-	118	-	-	139

Table 3.5 Full and reduced logistic regression height model estimates, chi-square test results, and fit statistics for models predicting one-year survival of Chinese tallow across a seasonally flooded bottomland hardwood forest at Boggy Slough Conservation Area. Variable information included in Table 3.3.

	Parameter		Wald			Fit Statistics		
Model/variable	**estimate**	**SE**	**Chi-Square**	***P*-value**	**Odds ratio**	**AIC**	**-2LL**	**ΔAIC**
Full - Height [a]						207.2	185.2	
Intercept	-0.68	1.31	0.27	0.601	0.51			
Height	0.10	0.03	13.53	0.000*	1.11			
Vigor (1)[b]	0.95	0.49	3.73	0.054*	2.58			
Vigor (3)	-0.17	0.55	0.10	0.756	0.84			
DWD (Yes)	-1.23	0.87	2.03	0.154	0.29			
M.T. (Depression)	-0.60	0.51	1.38	0.240	0.55			
M.T. (Raised)	0.99	0.55	3.21	0.073*	2.69			
M.T. (Slope)	-1.23	0.53	5.37	0.020*	0.29			
Canopy cover	-0.01	0.01	1.03	0.309	0.99			
1-TRT	0.74	0.51	2.12	0.146	2.10			
2-TRT	2.00	0.53	14.09	0.000*	7.40			
Reduced - Height						206.7	188.7	0.5
Intercept	-2.00	0.51	15.36	<0.0001*	0.14			
Height	0.10	0.03	13.78	0.000*	1.11			
Vigor (1)	0.81	0.48	2.89	0.089*	2.25			
Vigor (3)	-0.42	0.53	0.62	0.429	0.66			
M.T. (Depression)	-0.77	0.49	2.47	0.116	0.46			
M.T. (Raised)	1.00	0.55	3.29	0.070*	2.73			
M.T. (Slope)	-1.18	0.52	5.15	0.023*	0.31			
1-TRT	0.87	0.50	2.95	0.086*	2.38			
2-TRT	2.05	0.53	14.78	0.000*	7.78			

[a] Separate full and reduced models were estimated for height and diameter due to an inherently high Pearson's correlation (r = 0.9).

[b] Reference categories (in parentheses) represented in the intercept are medium stem vigor, no proximity to DWD, level ground, and an untreated area.
* denotes significance at $\alpha = 0.10$

Table 3.6 Full and reduced logistic regression diameter model estimates, chi-square test results, and fit statistics for models predicting one-year survival of Chinese tallow across a seasonally flooded bottomland hardwood forest at Boggy Slough Conservation Area. Variable information included in Table 3.3.

	Parameter		Wald		Fit Statistics			
Model/variable	**estimate**	**SE**	**Chi-Square**	***P*-value**	**Odds ratio**	**AIC**	**-2LL**	**ΔAIC**
Full - Diameter [a]						214.0	192.0	
Intercept	-1.15	1.32	0.76	0.384	0.32			
Diameter	0.45	0.16	7.94	0.005*	1.58			
Vigor (1)[b]	0.82	0.48	2.96	0.085*	2.27			
Vigor (3)	-0.10	0.54	0.04	0.850	0.90			
DWD (Yes)	-1.36	0.86	2.51	0.113	0.26			
M.T. (Depression)	-0.50	0.50	1.01	0.314	0.61			
M.T. (Raised)	1.05	0.54	3.75	0.053*	2.86			
M.T. (Slope)	-0.93	0.49	3.56	0.059*	0.40			
Canopy cover	-0.01	0.01	0.17	0.683	0.99			
1-TRT	0.54	0.49	1.23	0.267	1.72			
2-TRT	1.79	0.51	12.46	0.000*	6.00			
Reduced - Diameter						212.2	198.2	1.8
Intercept	-1.31	0.38	11.70	0.001*	0.27			
Diameter	0.39	0.15	7.10	0.008*	1.48			
M.T. (Depression)	-0.61	0.47	1.64	0.201	0.55			
M.T. (Raised)	1.04	0.53	3.83	0.050*	2.82			
M.T. (Slope)	-0.75	0.47	2.53	0.112	0.47			
1-TRT	0.38	0.45	0.72	0.395	1.47			
2-TRT	1.46	0.45	10.76	0.001*	4.32			

[a] Separate full and reduced models were estimated for height and diameter due to an inherently high Pearson's correlation (r = 0.9).

[b] Reference categories (in parentheses) represented in the intercept are medium stem vigor, no proximity to DWD, level ground, and an untreated area.
* denotes significance at $\alpha = 0.10$

Table 3.7 Full and reduced logistic regression height model estimates, chi-square test results, and fit statistics for models predicting one-year survival of native hypogeal species across a seasonally flooded bottomland hardwood forest at Boggy Slough Conservation Area. Variable information included in Table 3.3.

	Parameter		Wald			Fit Statistics		
Model/variable	**estimate**	**SE**	**Chi-Square**	***P*-value**	**Odds ratio**	**AIC**	**-2LL**	**ΔAIC**
Full - Height [a]						113.77	93.77	
Intercept	5.52	2.65	4.35	0.037*	250.49			
Height	0.30	0.13	4.97	0.026*	1.35			
Vigor (1)[b]	-1.36	0.61	4.97	0.026*	0.26			
Vigor (3)	-1.37	0.82	2.82	0.093*	0.25			
DWD (Yes)	-1.68	0.60	7.77	0.005*	0.19			
M.T. (Depression)	-1.19	0.74	2.54	0.111	0.31			
M.T. (Raised)	0.43	0.92	0.22	0.643	1.53			
M.T. (Slope)	-0.04	0.70	0.00	0.950	0.96			
Canopy cover	-0.04	0.03	2.21	0.137	0.96			
Weak flood tol.	0.59	0.58	1.01	0.315	1.80			
Reduced - Height						110.6	100.6	3.17
Intercept	1.56	0.56	7.78	0.005*	4.78			
Height	0.38	0.15	6.03	0.014*	1.46			
Vigor (1)	-1.26	0.56	5.02	0.025*	0.28			
Vigor (3)	-1.58	0.77	4.27	0.039*	0.21			
DWD (Yes)	-1.48	0.54	7.55	0.006*	0.23			

[a] Separate full and reduced models were estimated for height and diameter due to an inherently high Pearson's correlation (r = 0.9).
[b] Reference categories (in parentheses) represented in the intercept are medium stem vigor, no proximity to DWD, level ground, and moderate flood tolerance.
* denotes significance at $\alpha = 0.10$

Table 3.8 Full and reduced logistic regression diameter model estimates, chi-square test results, and fit statistics for models predicting one-year survival of native hypogeal species across a seasonally flooded bottomland hardwood forest at Boggy Slough Conservation Area. Variable information included in Table 3.3.

Model/variable	Parameter estimate	SE	Wald Chi-Square	*P*-value	Odds ratio	Fit Statistics AIC	-2LL	ΔAIC
Full - Diameter [a]						115.0	95.0	
Intercept	5.50	2.63	4.36	0.037*	244.87			
Diameter	1.93	0.86	5.09	0.024*	6.90			
Vigor (1)[b]	-1.34	0.61	4.81	0.028*	0.26			
Vigor (3)	-1.35	0.81	2.79	0.095*	0.26			
DWD (Yes)	-1.69	0.60	7.88	0.005*	0.19			
M.T. (Depression)	-1.22	0.74	2.74	0.098*	0.30			
M.T. (Raised)	0.41	0.91	0.20	0.651	1.51			
M.T. (Slope)	-0.13	0.70	0.03	0.857	0.88			
Canopy cover	-0.04	0.03	2.16	0.142	0.96			
Weak flood tol.	0.70	0.57	1.49	0.222	2.02			
Reduced - Diameter						113.0	103.0	2.0
Intercept	1.65	0.57	8.45	0.004*	5.23			
Diameter	2.37	1.02	5.44	0.020*	10.69			
Vigor (1)	-1.23	0.56	4.81	0.028*	0.29			
Vigor (3)	-1.59	0.75	4.44	0.035*	0.21			
DWD (Yes)	-1.49	0.54	7.73	0.005*	0.23			

[a] Separate full and reduced models were estimated for height and diameter due to an inherently high Pearson's correlation (r = 0.9).
[b] Reference categories (in parentheses) represented in the intercept are medium stem vigor, no proximity to DWD, level ground, and moderate flood tolerance.
* denotes significance at $\alpha = 0.10$

Table 3.9 Full and reduced logistic regression height model estimates, chi-square test results, and fit statistics for models predicting one-year survival of native epigeal species across a seasonally flooded bottomland hardwood forest at Boggy Slough Conservation Area. Variable information included in Table 3.3.

	Parameter		Wald			Fit Statistics		
Model/variable	**estimate**	**SE**	**Chi-Square**	***P*-value**	**Odds ratio**	**AIC**	**-2LL**	**ΔAIC**
Full - Height [a]						133.43	113.43	
Intercept	3.67	2.48	2.18	0.140				
Height	0.36	0.12	8.60	0.003*	1.44			
Vigor (1)[b]	-0.19	0.57	0.11	0.737	0.83			
Vigor (3)	-0.84	0.76	1.23	0.267	0.43			
DWD (Yes)	-0.12	0.83	0.02	0.890	0.89			
M.T. (Depression)	-0.20	0.60	0.11	0.742	0.82			
M.T. (Slope)	-0.59	0.61	0.93	0.335	0.55			
Canopy cover	-0.04	0.03	1.83	0.176	0.97			
Shade intolerant	0.36	0.59	0.38	0.539	1.44			
Shade tolerant	0.59	0.64	0.83	0.361	1.80			
Reduced - Height						123.5	117.5	9.93
Intercept	4.05	2.44	2.76	0.096*				
Height	0.32	0.11	9.19	0.002*	1.38			
Canopy cover	-0.04	0.03	2.42	0.120	0.96			

[a] Separate full and reduced models were estimated for height and diameter due to an inherently high Pearson's correlation (r = 0.9).
[b] Reference categories (in parentheses) represented in the intercept are medium stem vigor, no proximity to DWD, level ground, and very shade tolerant.
* denotes significance at $\alpha = 0.10$

Table 3.10 Full and reduced logistic regression diameter model estimates, chi-square test results, and fit statistics for models predicting one-year survival of native epigeal species across a seasonally flooded bottomland hardwood forest at Boggy Slough Conservation Area. Variable information included in Table 3.3.

	Parameter		Wald		Fit Statistics			
Model/variable	**estimate**	**SE**	**Chi-Square**	***P*-value**	**Odds ratio**	**AIC**	**-2LL**	**ΔAIC**
Full - Diameter [a]						136.7	116.7	
Intercept	4.07	2.44	2.78	0.095*				
Diameter	2.28	0.83	7.63	0.006*	9.78			
Vigor (1)[b]	-0.23	0.56	0.17	0.679	0.79			
Vigor (3)	-0.77	0.75	1.06	0.303	0.46			
DWD (Yes)	-0.16	0.79	0.04	0.838	0.85			
M.T. (Depression)	-0.20	0.60	0.11	0.741	0.82			
M.T. (Slope)	-0.55	0.60	0.85	0.356	0.57			
Canopy cover	-0.04	0.03	2.14	0.144	0.96			
Shade intolerant	0.38	0.58	0.41	0.520	1.46			
Shade tolerant	0.46	0.63	0.54	0.462	1.59			
Reduced - Diameter						126.0	120.0	10.7
Intercept	4.40	2.40	3.35	0.067*				
Diameter	2.08	0.73	8.02	0.005*	8.00			
Canopy cover	-0.04	0.03	2.77	0.096*	0.96			

[a] Separate full and reduced models were estimated for height and diameter due to an inherently high Pearson's correlation (r = 0.9).
[b] Reference categories (in parentheses) represented in the intercept are medium stem vigor, no proximity to DWD, level ground, and very shade tolerant.
* denotes significance at $\alpha = 0.10$

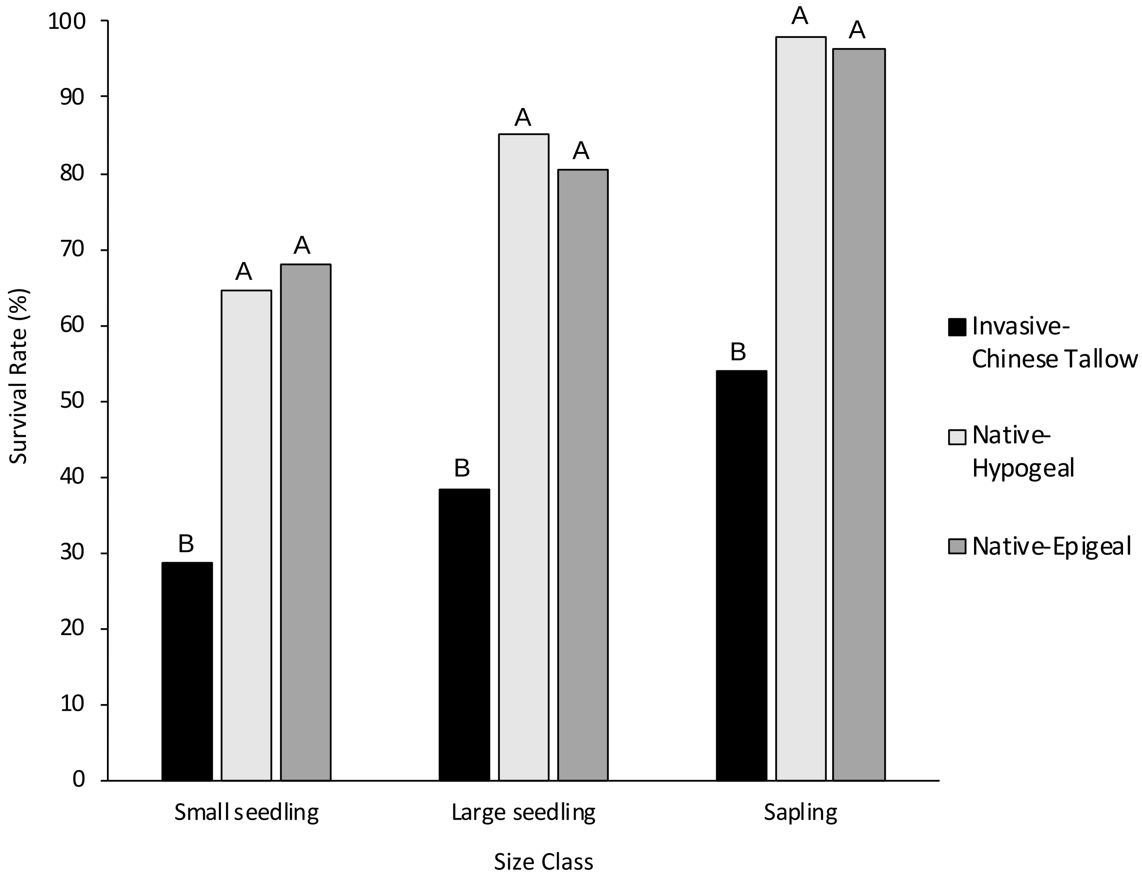

Figure 3.1 Proportions of survival for invasive-Chinese tallow, native-hypogeal, and native-epigeal species groups among three sizes classes: small seedling (0.5 to 2 ft. tall), large seedling (>2 ft. tall, but <0.6 inches at DBH), and sapling (0.6 to 3.9 inches DBH). Letters represent significant differences ($p \leq 0.10$) among each size class from chi-square tests for association.

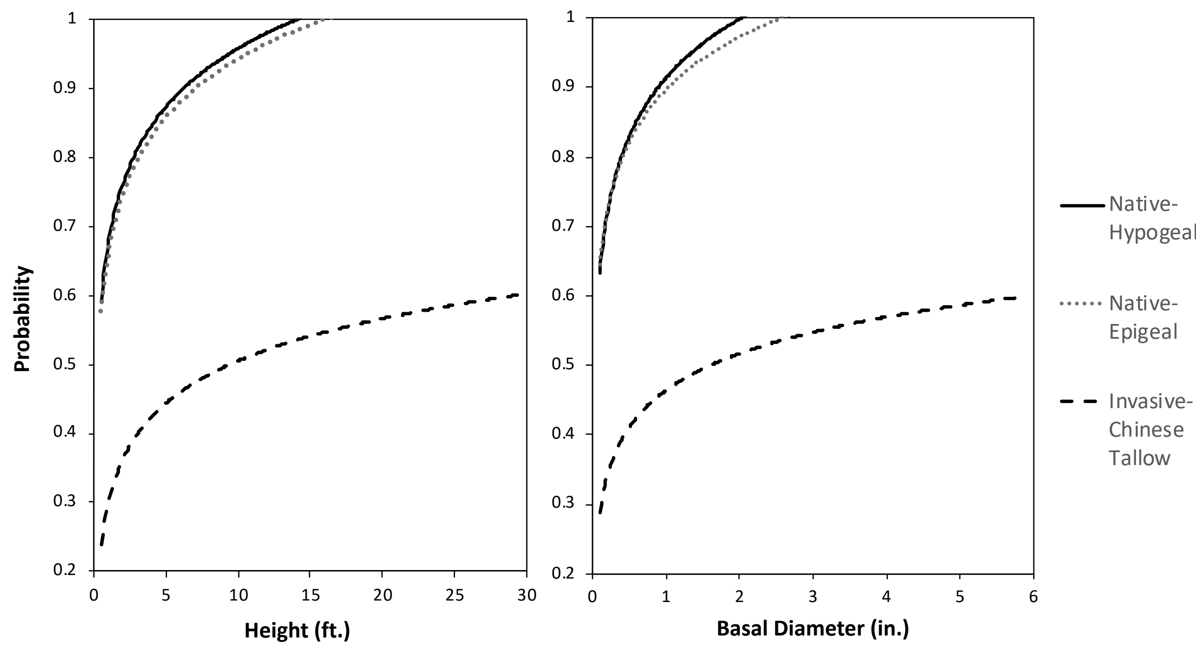

Figure 3.2 Predicted probabilities of native-hypogeal, native-epigeal, and invasive-Chinese tallow species survival from reduced height (left) and reduced diameter (right) logistic regression models as stem size increases.

Table 3.11 Gauge height (feet) and flooding data for the Neches River from the USGS monitoring station near Diboll, TX (ID #: 08033000), approximately thirteen miles south of Boggy Slough Conservation Area.

	08/01/2020 – 07/31/2021		
Month	**Avg. Gauge Height (ft.)**	**# of Days Above Action Stage [a]**	**Historical Avg. Gauge Height (ft.) [b]**
August	2.2	0	2.9
September	4.7	0	2.8
October	2.8	0	3.6
November	2.3	0	5.7
December	4.1	0	7.4
January	10.9	9	9.4
February	9.1	1	9.5
March	10.2	0	9.4
April	8.9	1	9.0
May	14.8	31	7.5
June	12.5	20	7.1
July	9.1	7	4.5

[a] The action stage (11.5 feet) represents the level where some type of mitigation action must be taken in preparation for possible significant hydrologic activity. Bankfull stage data were not available at this location, but would occur at a lower gauge height.
[b] Means calculated from data collected between October 1990 and July 2020.

Chapter IV: Conclusions and Management Implications

In many riparian forests, natural regeneration of desired, native species can struggle to establish and survive due to compounded effects from a variety of threats. These include land loss/conversion, histories of high-grading, anthropogenic hydrological regime changes, and encroachment of exotic species such as Chinese tallow. Healthy, functioning floodplains are dependent on a diverse array of native tree species that play a role in many critical ecosystem services. Bottomland forests sequester carbon, support biodiversity, purify water sources, control flooding and erosion, and more. This research furthered the understanding of the status of desired, native tree regeneration in bottomland hardwood forests through assessment of native and invasive dynamics and survival.

This study investigated how species composition varied in the aftermath of previous efforts to control invasive species. Differences in species abundance were examined among two areas that had herbicide treatments of Chinese tallow and one untreated area. Densities of several species in small and large seedling classes were found to vary in the post-treatment environments, though time since treatment was likely not great enough to observe many effects in the sapling layer. Native species that were indicators in the untreated area were generally shade-tolerant, with the exception of overcup oak. Opportunistic, late-seral native species arose as indicators in the treated areas, along with Chinese tallow. The post-treatment environment appeared to facilitate new cohorts of tallow regeneration, although future monitoring will be necessary to see if these effects persist into the sapling stage. Based on these results, efforts to control tallow may be more effective in phases, with a second delayed application to decrease abundance of new seedlings a year or two after the initial treatment. Management initiatives in these systems that aim to preserve the historical overstory composition must provide opportunities for recruitment of desired intolerant species, such as oaks, without allowing fast-growing invasive species to take advantage of newly available resources first. Regeneration methods such as shelterwood or group selection may be beneficial for increasing density of these species, as they increase light availability but create less extreme of an environment than would a clearcut. Cleaning and weeding are also important intermediate methods to control less desired or invasive species. Future research projects could compare regeneration data from this study site, which has not undergone any harvesting methods, to sites where more rigorous regeneration harvest methods have taken place in order to assess the influence of management on regeneration success.

This research also examined the relationships between native regeneration, abiotic, and biotic factors in order to identify where desired species may occur most abundantly. Seedlings and saplings were correlated with complex light and ground cover gradients, as well as microtopography. For most species, these relationships varied across small seedling, large seedling, and sapling size classes. It is crucial to match species to site when managing for desired regeneration, both in terms of determining the most effective management methods and/or selecting species for artificial regeneration. Additionally, changes in a species' optimal conditions over time should be considered. These results provided a snapshot of dynamics in the late growing season of one year; future research spanning changes in conditions throughout a year and across multiple years would also further the understanding of complex regeneration ecology in these systems.

One-year survival was evaluated in order to further understand how stem characteristics and micro-site conditions may impact regeneration success and future dominance probability. Chinese tallow survival was lower than that of native species groups, which may have been due to snow and cold temperatures in February 2021. Height and diameter were influential predictors of survival for all species groups. Other factors, such as vigor, microtopography, canopy, and proximity to DWD, provided information on potential group-specific, short-term filters to regeneration and factors to consider when implementing management plans. Future long-term research would be beneficial to better understand future forest conditions, the relationship between inherent site factors and climatic patterns, and implications of tallow mortality. Overall, this research provided a closer look at some of the complex dynamics in bottomland hardwood forests invaded by Chinese tallow. This information can be utilized to understand the condition of these systems in the face of anthropogenic and biotic threats and provide insight about projected species composition in the future and potential management actions.

VITA

Lydia Voth Rurup received her Bachelor of Science degree from the University of Minnesota–Twin Cities in May 2019, with a major in Forest and Natural Resource Management and specialization in Forest Ecosystem Management and Conservation. She also minored in Urban and Community Forestry and Environmental Science, Policy, and Management. During her time at UMN, she enjoyed working as an undergraduate research assistant in two different labs, which helped lead to her decision to pursue a graduate degree. Lydia began graduate school at Stephen F. Austin State University in January 2020, and graduated with a Master of Science degree in Forestry in December 2021. Under the guidance of Dr. Rebecca Kidd, she conducted research on floodplain forest regeneration dynamics and survival in the face of invasive species at Boggy Slough Conservation Area. While at SFASU, she was a graduate teaching assistant for the undergrad Forest Ecology course and also aided other research projects.

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This thesis was typed by Lydia Voth Rurup, using the Stephen F. Austin State University Graduate School Style Manual.