Summer 8-10-2024

Chinese Tallow Long-Term Impact on Stand Dynamics in a Bottomland Hardwood Forest Following Vegetation Management

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Presented to the Faculty of the Graduate School of

Stephen F. Austin State University

In Partial Fulfillment

Of the Requirements

For the Degree of

Masters of Science in Environmental Science

Division of Environmental Science

STEPHEN F. AUSTIN STATE UNIVERSITY

August 2024

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Abstract

Chinese tallow (Triadica sebifera (L.) Small) is an invasive tree known to have negative impacts on bottomland hardwood forests. Chinese tallow is proficient in reproducing and surviving in bottomland hardwood forests because it is more flood and shade tolerant than most native trees. The long-term effectiveness of Chinese tallow control is an important topic to research as only a few long-term studies of Chinese tallow control have been conducted. A 10-year re-measurement of 22 paired plots installed in 2012 on the Pineywoods Mitigation Bank near Diboll, Texas following herbicide treatments examined Chinese tallow and native tree stand structure. Each native and Chinese tallow plot included three sets of nested subplots. The nested subplots include one overstory, one sapling, and four seedling subplots. R (v4.3.3) software was used to run repeated measures analysis of variance (ANOVA), simple linear regressions, and other summary statistics. Chinese tallow density did not differ between years and herbicide treated and non-herbicide treated plots ($p = 0.414$). Relative density of Chinese tallow had a correlation between years and herbicide treated and non-herbicide treated plots ($p = 0.040$). Chinese tallow had a significant decrease in herbicide treated plots from 2012-2013 to 2022 (81.46% decrease from 2012). Native stand structure metrics did not differ between years and herbicide treated and non-herbicide treated plots (density $p = 0.883$, basal area $p = 0.843$, quadratic mean diameter $p = 0.851$, relative density $p = 0.901$).

Diversity and evenness differed in all sapling plots from 2022 data to 2012-2013 data (diversity 2022-2012 p = <0.001, 2022-2013 p = <0.001). The change in overstory tree density was not correlated with years since herbicide treatment (density p = 0.797, basal area p = 0.335, quadratic mean diameter p = 0.544, stand density index p = 0.272). This suggests that there is no difference in stand metrics between 6 and 10 years. Observational study on top-killed Chinese tallow likely killed from the 2021 freeze that occurred in East Texas showed that only 12.4% of Chinese tallow were top-killed from the freeze. All of the top-killed Chinese tallow were observed having sprouts growing out of the snag. There was no correlation between snag diameter at breast height and maximum sprout height (p = 0.172). I found no support for my hypotheses that Chinese tallow and native tree densities responded to herbicide application. Overall forest structure also did not respond to herbicide, although I found that relative density of only Chinese tallow was affected in Chinese tallow plots treated with herbicide. It was also found that the number of years since the last herbicide treatment did not correlate to the magnitude of changes in Chinese tallow and native tree densities. The observational study on top killed Chinese tallow from the 2021 freeze showed no correlation between maximum seedling sprout height relating to tree diameter at breast height (dbh). With updated 10-year maximum stand density index results we found that stand density index in bottomland hardwoods is substantially greater than upland hardwood stands reported in the central hardwoods region (Schnur 1937). Our results follow the pattern of another long-term Chinese tallow control study (Norman 2020), where change in Chinese tallow density was not correlated with the treatment type. Reduction in diversity metrics followed the same trend from invasive plant studies (Hart and Holmes 2013; Hejda et al. 2009; Zedler and Kercher 2004) that biodiversity will reduce over time when an area is invaded. Future long-term studies are needed to provide important information regarding biodiversity reduction to land managers

Acknowledgments

There are many people who I would like to thank for their role in the completion of this thesis. First of all, I would like to thank Dr. Stovall for taking me on a year into graduate school and guiding me through the process. I would like to thank my committee members, Dr. Kidd, Dr. Weng, and Dr. Rashall for their help and input. Thanks to Dr. Farrish for his encouragement and advice on graduate school and the professional field. Thanks to Cliff Sunda and Bo Reid for giving me access to the Piney Woods Mitigation Bank.

A special thanks to Martin Hadsell, Paige Graham, and Clifton Albrecht for their help in conducting fieldwork. I wouldn't have been able to collect the data if it wasn't for them enduring cold weather, wet shoes, stuck vehicles, and long hikes. Another special thanks goes out to Connor Adams for his help. Without his help this thesis wouldn't be in as good of shape as it is. I would also like to thank all of my friends in graduate school for their help, input, and support that I needed. A special thanks to my coworkers at Mitigation Resources of North America as well for all of their great advice and the support they gave me to finish my graduate degree.

And finally, I couldn't have done this without the support of my family. I really appreciate your prayers and guidance. Thanks for pushing me to finish what I started and

never letting anything let me lose focus of my goal of obtaining my graduate degree. I love you guys so much.

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Introduction

Biological invasions, along with pollution and habitat change, are a major contributor to the decline of biodiversity throughout the world (Pysek and Richardson 2010). Chinese tallow (*Triadica sebifera* (L.) Small) is a non-native species that is invading bottomland hardwood forests in the southeastern United States. Chinese tallow is proficient in reproducing and surviving in many ecosystems in this region because it comes from similar latitudes and ecosystems in China (Jubinsky and Anderson 1996). Many land managers focus on combating the impact that Chinese tallow has on native habitats.

Bottomland hardwood forests, with an abundance of native trees and are adapted to flooding, can be affected greatly by the invasion of tallow. Chinese tallow can outcompete native species, like oaks, during natural regeneration processes because of high seed production (Yang et al. 2021). The mesic environments of bottomland hardwood forests are more suitable for Chinese tallow growth than other environments, as Chinese tallow is tolerant to frequent flooding and drought that occurs in these bottomlands (Nolte 2013). Managing for Chinese tallow is a difficult endeavor. Because of Chinese tallow's tolerance to stressful conditions and its ability to suppress fire (Pile et al. 2017), herbicides are commonly used to control Chinese tallow. The effectiveness of herbicides on long-term control of Chinese tallow is the subject of few studies (Enloe et al. 2015). Learning how to control Chinese tallow in bottomland hardwood forests is important in land management for wildlife, mitigation banks, and timber production.

Literature Review

Chinese Tallow

History

Chinese Tallow is a native deciduous tree in eastern Asia and has become a predominant invasive species in Southeastern United States forests (Pile et al. 2017). Chinese tallow originated from the Yangtze Delta in China and spread south and west following silk production and the development of waterways. The seeds were easy to transport and the waterways provided a good environment for them to grow and thrive. The wax from Chinese tallow was used to make ceremonial candles for the Buddhists around 581 to 618 AD. Buddhists refrain from killing animals because of their religion, so Chinese tallow provided a good substitute for animal source of tallow. Beyond the United States, Chinese tallow was also introduced in Japan, the Philippines, Taiwan, Northern Vietnam, and other areas of China with the help of Fujianese merchants. During the 1700's Chinese tallow spread into northern India and on islands in the Caribbean (Batchelor 2017).

Chinese tallow was first introduced into the U.S. in Savannah, Georgia, in 1772 and thought to have been first planted at Noble Jones's Wormsloe Plantation from seed supplied by Benjamin Franklin (Batchelor 2017). Chinese tallow was later documented in

South Carolina in the late 1700s (Pile et al. 2017). Another hypothesis suggests that the introduction of Chinese tallow in the United States may have occurred when Henry Laurens, a South Carolina rice farmer received seeds from John Blake, an English scientist, and shared the seeds with other farmers in the area for the purpose of making wax for candles (Batchelor 2017). In Texas and Louisiana, Chinese tallow introductions occurred in the early 1900s by the United States Department of Agriculture for the purpose of making soap, where it spread rapidly along the coastal and bottomland regions (Jamieson and Mckinney 1938). Chinese tallow is also used as an ornamental tree in many gardens and yards.

Though Chinese tallow does grow in similar ecosystems in the United States and China, the genotypes differ between the native grown Chinese tallow in China and the introduced Chinese tallow in the United States. Normally, invasive genotypes grow faster, have lower defenses, and have higher reproductive rates. This allows invasive plants, especially Chinese tallow, to outcompete and overcome native competitors in forested and prairie ecosystems (DeWalt et al. 2011). Invasive species adapting to local climate and ecological conditions allows the species to extend their range. The frequency and patterns of reproduction are important for the speed of local adaptation because the demographic and environmental conditions in which reproduction occurs may change (Barrett et al. 2008). The differences in genetically determined phenotypes for Chinese tallow were determined, where native genotypes of Chinese tallow from Taiwan had less basal area and higher concentrations of tannins than introduced genotypes of Chinese tallow in Southeastern United States (Siemann and Rogers 2001). Genotypes in Georgia, that came from some of the first introduced Chinese tallow have a greater allocation to defense and less growth than newer introduced genotypes in Texas and Louisiana. The genotypes from Georgia have gone through more evolution to the landscape than those in Texas and Louisiana since they were introduced a couple of centuries before (DeWalt et al. 2011). Studies have also found that Chinese tallow growing in Savannah, Georgia, and Charleston, South Carolina, are genetically more adapted to their new environments due to over a hundred years of evolution, suggesting that Chinese tallow was introduced into these areas of the United States first (DeWalt et al. 2011).

Ecology

Chinese tallow is classified as a facultative species in the Atlantic and Gulf Coastal Plains, meaning they can grow in both wet and upland sites (USDA 2023). Grasslands, bottomland hardwoods, and disturbed areas are at higher risk of being infested with Chinese tallow because it is easy for Chinese tallow to disperse seeds (Urbatsch 2018). The establishment and spread of Chinese tallow are due to their high seed production and ability to grow under a variety of conditions (Yang et al. 2021). Chinese tallow seedlings are shade tolerant and can grow in the sun rapidly. Seedlings are also highly tolerant to herbivory and plant competition (Gan et al. 2009). Studies have found that invasive Chinese tallow in the United States has higher flavonoids in the roots and in general than Chinese tallow in China (Luo et al. 2022). Flavonoids offer UV protection, resistance to drought, and resistance to fungal activity. Also, invasive Chinese tallow in the United States has lower concentrations of tannins than native populations in China. More birds to prefer Chinese tallow fruits over other native trees because of the low concentration of tannins helping spread Chinese tallow seeds faster than the natives (Wang et al. 2012).

A mature Chinese tallow can produce up to 310,000 seeds per year and has many pathways of seed dispersal, including streams and water bodies primarily, but also bird species, human activities like construction, and natural disturbances like hurricanes and flooding (Gan et al. 2009). Chinese tallow seed release usually starts in September and lasts until November, and the seed can survive for multiple years in soils until it germinates. Reproductive age of Chinese tallow can be as early as three years and persist for up to 100 years (Yang 2019).

Chinese tallow is a tetraploid, which is a trait that allows the tree to adapt quickly to changing environments because it has twice the number of chromosomes in each cell compared to diploid organisms (Gao et al. 2016). Tetraploid plants are found to have more growth vigor than diploids. Chinese tallow have adapted to have a high unsaturated fatty acid in their seeds, which attracts more birds to eat the fruits and eventually disperse more seeds (Luo et al. 2022). Chinese tallow has also adapted to have attractive reddish-orange leaf color in the fall and winter to attract more birds to the trees. Additionally, Chinese tallow were found to have genes that linked to resistance to disease, secondary metabolite biosynthesis, and nutrition energy utilization (Luo et al. 2022).

Climate and geography play a role in tallow establishment. The regional distribution of Chinese tallow is limited by elevation, as there is no Chinese tallow found higher than 540 feet above sea level or on a slope that is steeper than 18 degrees (Yang et al. 2021). Studies have found a correlation between the presence of Chinese tallow in both native and invaded ranges and bioclimatic factors (Gan et al.2009). In Pattison and Mack (2009), climate matching models were used to predict the spread of tallow north of its known range and found that seed germination and young plant performance were inhibited at northern and inland sites. They also found that the low light levels in closed canopy habitats negatively affected the Chinese tallow. Therefore, we can assume that Chinese tallow will unlikely spread out of the southern region of the United States.

Chinese tallow grows in clay, loam, and sandy soils that have a pH range of 3.9 to 8.5 (Lin et al. 1958). Fast growth rates of invasive species can advance soil processes and make nitrogen availability higher. Zou et al. (2006) performed a study to examine Chinese tallow's effects on soil nitrogen availability, shoot-specific respiration rates, and soil-plant system carbon and nitrogen cycling. They found that Chinese tallow has significantly greater shoot, root, and total mass than native species. These growth rates affected soil processes by having a lower amount of organic nitrogen than soil where native species grow. This is supported by the nitrogen mineralization rates being higher for soils that had Chinese tallow, which also leads to Chinese tallow soils having more inorganic nitrogen. The high availability of inorganic nitrogen may lead to Chinese tallows invasiveness because increased access to inorganic nitrogen increases the rate of plant growth.

Following disturbances in southern forests Chinese tallow will outcompete native trees, like loblolly pine (*Pinus taeda* L.) and water oak (*Quercus nigra* L.), because of its rapid height growth. Hurricanes can cause destruction and saltwater incursion into bottomland hardwood soils along the Gulf Coast. Most native plants can't tolerate elevated salinity in soils, but Chinese tallow is able to tolerate this stress and is not affected to the same degree (Chen et al. 2013). Forest types, like oak (*Quercus* spp.)/gum (*Nyssa* spp.)/cypress (*Taxodium distichum* (L.) Rich.) and elm (*Ulmus* spp.)/ash (*Fraxinus* spp.)/cottonwood (*Populus deltoides* W. Bartram ex Marshall) have the highest amount and rate of spread of Chinese tallow among the forest types in East Texas. These habitats are found in bottomlands and lowlands where seed dispersal by water establishes Chinese tallow faster and in a wider range. Chinese tallow is less likely to establish in sloping uplands where various pine species grow because of the limited amount of seed dispersal (Fan et al. 2012). Already established native forests can over time lose all of the native species and be overrun by Chinese tallow because native seedlings can be outcompeted by Chinese tallow.

Chinese tallow is exceptionally tolerant to flooding, which is a common occurrence in bottomland hardwoods. Species that are flood tolerant tend to lose the primary root system and grow succulent soil water roots (Conner 1994). Chinese tallow has been found to have a greater reduction in biomass of its roots when flooded. Hypertrophied lenticels form on Chinese tallow when flooded to help avoid the stress from the lack of oxygen from flooding. Chinese tallow seedlings can outlast cypress seedlings in flooded conditions, and cypress trees are one of the best native species at surviving in flooded soils (Conner 1994).

Chinese tallow spread rapidly throughout the Southeastern U.S. occupying around 185,000 ha of southern forests by 2008 (Oswalt 2010). In 2007 Chinese tallow became the fourth most common tree in Southeast Texas behind loblolly pine, sweetgum (*Liquidambar styraciflua* L.), and water oak (Oswalt 2010). The fifth most common tree in Louisiana is Chinese tallow, after loblolly pine, sweetgum, red maple (*Acer rubrum* L.), and water oak. From 1991 to 2005 Chinese tallow population increased in Louisiana by more than 500 %. Mississippi had a 445 percent increase of Chinese tallow from 1994 to 2006. East Texas only had a 174 percent increase of Chinese tallow. The 2005 study in Louisiana found 689 ± 159 million Chinese tallow seedlings, mostly located in the southwest, south delta, and southeast areas of the state. A survey in 2006 conducted in Mississippi found 104 ± 50 million Chinese tallow seedlings located in the south and southwest areas of the state. The 2007 survey conducted in Texas found 931 ± 295 million Chinese tallow seedlings with the majority in the southeast area of the state

(Oswalt 2010). Louisiana and Mississippi had around a six percent increase of Chinese tallow saplings in all diameter classes, while the number in each diameter class in Texas tripled in under ten years. East Texas Chinese tallow volume increased by 66 million cubic feet in under ten years, while Mississippi had a 23 million cubic feet increase in around the same time period. Louisiana had a 395 percent increase in Chinese tallow volume in under ten years (Oswalt 2010).

Control Methods

One of the most effective methods of controlling Chinese tallow is the use of herbicides. Herbicide may be applied to Chinese tallow through basal bark, foliar, frill, and girdle techniques (Jubinsky and Anderson 1996). It is recommended for large Chinese tallow trees to make stem injections using diluted imazapyr herbicides (Miller et al. 2013). Felled trees and cut saplings often have triclopyr herbicide applied to the tops of the stumps and sides immediately after cutting using a spray bottle, brush, or backpack sprayer. For extensive forest infestations, a hexazinone herbicide can be applied to the soil surface within three feet of the stem. Chinese tallow saplings can be sprayed on the bark and basal area with a triclopyr herbicide mixed with water and a surfactant like mineral oil or diesel fuel. Chinese tallow seedlings should be thoroughly sprayed with any imazapyr or triclopyr herbicides mixed with water and a surfactant (Miller et al. 2013). The best time of the year to apply foliar application of herbicide to Chinese tallow is in the fall. This is because the tree is moving its nonstructural carbohydrates from the foliage to its root system, increasing the chance that the herbicide is transported to the roots, eventually killing the tree (Conway et al. 1999).

In newly planted bottomland areas, weeds and vines can overtake and outcompete native tree seedlings. Specific herbicides can be used to control the herbaceous vegetation without killing the trees, increasing the survivability of oaks by 25 percent and other species like sweetgum by 10 to 15 percent (Stanturf et al. 2004). It is recommended that herbicide treatment should be done three to five years after clear-cutting or planting (Nix 2004). After native trees have been established or when managing a productive forest, invasive species can be controlled using stem injections and backpack spraying of herbicide

The most common herbicides used for controlling Chinese tallow are imazapyr and triclopyr. Imazapyr is a main ingredient in broad-spectrum herbicides mostly used in non-agriculture practice, such as forestry. The function of imazapyr is to control perennial weeds, shrubs, and non-crop trees pre- and post-emergence (Douglass et al. 2016). Plants absorb imazapyr through the roots and foliage and it spreads rapidly through the xylem and phloem and accumulates in the meristematic regions. Imazapyr kills plants by inhibiting acetohydroxyacid synthase, which is involved in the biosynthesis of branched-chain essential amino acids. The disruption of this process interferes with DNA synthesis and stops cell growth in the plant (Roberts et al. 2007).

Like imazapyr, triclopyr is a main ingredient used in herbicides to control woody plants and broad-leaved weeds for the preparation of planting trees. Triclopyr is an auxin mimic herbicide, meaning that plants are killed by triclopyr mimicking the plant growth hormone auxin (Tu et al. 2001). By inhibiting the indole acetic acid, known as auxin, the plant will grow in an uncontrolled and disorganized way eventually leading to its death. Although not fully understood, the uncontrolled growth of the plant is caused when triclopyr acidifies and loosens cell walls, resulting in cells expanding without control (Tu et al. 2001). Triclopyr can be either adsorbed by the plant foliage or its roots in the soil. Four factors for triclopyr to be successful for woody plant control include herbicide absorption, photosynthesis for movement of the herbicide, herbicide translocation, and meristematic activity (Radosevich 1979).

Root sprouting, collar sprouting, and lateral sprouting are responses of Chinese tallow to disturbance. Sprouting can be caused by herbicide application, fire, mastication, and freezes. Failed control efforts often lead to Chinese tallow sprouting from roots or stump (Norman 2020). Enloe et al. (2015) suggests that when using triclopyr, Chinese tallow root sprouted and triclopyr did not produce consistent control of tallow. Aminocyclopyrachlor was found to have better results than triclopyr by reducing foliar cover, lateral root sprouting, and root collar sprouting. The study also found several active ingredients in herbicides that work as well or better than triclopyr, including fluroxypyr, aminopyralid, and imazamox (Enloe et al. 2015).

Prescribed fire is another management tool, but has not been as effective in controlling Chinese tallow (Grace 1998). Chinese tallow has adapted to surviving and even suppressing fires. Large Chinese tallow has thicker bark that keeps the tree from being killed. Smaller Chinese tallow that is top-killed from fire often resprouts from the roots. Chinese tallow only catches fire in the crown from only the hottest of fires (Grace 1998). Frequent fires can increase the risk of Chinese tallow invasion because the fire may eliminate understory herbaceous vegetation and provide optimal conditions for Chinese tallow seeds to germinate and grow (Cheng et al. 2021).

Mastication is another method used to control Chinese tallow by mechanically shredding the trees to ground level (Donahue et al. 2004). Once cut down, Chinese tallow stumps are sprayed with herbicide. This method encourages native plant regeneration by controlling Chinese tallow regeneration. Applying this method in the spring is best because Chinese tallow seeds will not be dispersed once the tree is shredded (Donahue et al. 2004; Pile et al. 2017).

Bottomland Hardwoods

Bottomland Hardwood Forest Landscape

Occurring on alluvial floodplain sites along rivers and streams, bottomland hardwood forests are found extensively throughout the southeastern United States. These forests form on the easily eroded, poorly consolidated sedimentary soils of the East and West Gulf Coastal Plains, the Atlantic Coastal Plain, and the Mississippi Alluvial Plain

(Hodges 1997). Over time, meandering of the rivers and streams creates the topography of the forests and leaves natural levees, swamps, oxbow lakes, and ridges that make up the ecosystem of bottomland hardwood forests. Flooding is common in the wintertime when fewer plants are taking up water and rain is frequent. Although bottomland hardwood forest topography is typically relatively flat, there are slight changes in elevation by a couple of feet. Flooding events can erode soil and displace it. (Oswalt et al. 2016).

Natural levees are created when overbank flow suspends sediment and deposits them adjacent to channels. The levees form mostly on concave banks or straight reaches and can range from 30 meters to 100 meters wide. The soil content of natural levees consists of sand and silt particles from the stream channel. Flood basins and swamps occur behind the natural levees and are relatively flat with almost no changes in elevation. These flats and swamps hold flood water semi-permanently to permanently and include finer textured soils. Oxbow lakes occur on the flats and form when a meandering stream channel is cut off due to erosion. Ridges are formed on point bars when flood water recedes leaving the sediment behind. Ridges are higher than natural levees and can create permanently flooded depressions. The ridges are mostly composed of coarse textured sediments and vegetation will quickly form communities stabilizing the ridges (Wharton 1982).

Bottomland Harwood Forest Climate

Bottomland hardwood forests receive an average of 1270 mm of rainfall per year. Usually, these forests will have a dry season from August to October. In the southern part of the region where bottomland hardwood forests are found, the average temperature is approximately be around 21 ℃. Conditions in the areas are mostly humid throughout the year leading to high rates of decomposition. The growing season for these forests is long, ranging from 240 to 320 days. The growing season is also known as the "frost-free" period (Mundorff 1998).

Nutrient Cycling

Bottomland hardwood forests are highly productive for vegetation growth because of the rotation of nutrients brought in by sediment deposition. Floodwaters bring nutrients into the plant communities where it settles into the soil to later be taken up by the plants. Any unused nutrients are then transported back to the river channel where they will be available for plant communities downstream (Wharton 1982). These forests also can act as a filter for nutrients flowing from adjacent lands into the rivers. Excess inorganic nitrogen and phosphorus can cause eutrophication in water bodies such as rivers and lakes. The plant community in bottomland forests slows the flow of inorganic nitrogen by using it and assisting the nitrogen cycle to reduce the amount deposited into the river (Devito and Dillon 1993; Xu 2006). Bottomland forests essentially act as a buffer partially trapping the nutrients from flowing downstream and depositing into the ocean (Wharton 1982).

Bottomland Hardwood Forest Vegetation

Vegetation in bottomland hardwood forests is made up of diverse trees and herbaceous vegetation that are adapted to the ecosystem of floodplains. The most flooded parts of the forest are the lowest parts and form swamps that include cypress-tupelo-gum species and black willows (*Salix nigra* Marshall) as well. In the semi-permanently flooded soils of slightly higher elevations species like black willow, overcup oak (*Quercus lyrata* Walter), water hickory (*Carya aquatica* (Michx. F.) Nutt.), green ash (*Fraxinus pennsylvanica* Marshall), red maple, and river birch (*Betula nigra* L.) are found. Areas that are flooded for up to two months during the growing season include a wider variety of tree species like laurel oak (*Quercus laurifolia* Michx.), persimmon (*Diospyros virginiana* L.), green ash, American elm (*Ulmus americana* L.), sweetgum, sugarberry (*Celtis laevigata* Willd.), red maple, willow oak (*Quercus phellos* L.), and sycamore (*Platanus occidentalis* L.). Areas that are still on the floodplain but are infrequently flooded, usually less than a week to a month at times, are dominated by oaks, such as swamp chestnut oak (*Quercus michauxii* Nutt.), cherrybark oak (*Quercus pagoda* Raf.), water oak, and several hickories (Mitsch and Gosselink 2015; Diggs et al. 2006). Trees mentioned above are not strictly bound to the zones they are usually found in; they can be spread throughout because of the changing landscape over time and flooding.

Other vegetation found in bottomland hardwoods includes shrubs and vines. Some common shrubs found in East Texas bottomland hardwoods are possumhaw (*Ilex decidua* Walter), green hawthorn (*Crataegus viridis* L.), and swamp privet (*Forestiera acuminata* (Mich.) Poir.). Common vines found growing in this ecosystem are greenbrier (*Smilax* spp.), American buckwheat vine (*Brunnichia ovata* (Walter) Shinners), poison ivy (*Toxicodendron radicans* (L.) Kuntze), peppervine (*Nekemias arborea* (L.) J. Wen & Boggan), and trumpet creeper (*Campsis radicans* (L.) Seem. Ex Bureau) (Diggs et al. 2006).

Vegetation Adaptations

Most of these species are considered flood tolerant and even have adapted to flooding and being waterlogged for long periods of time. Adventitious roots are common in black willows that grow in flooded areas of bottomlands. These roots form when there is a concentration of ethylene in hypoxic tissues and assist the tree in obtaining oxygen in its root systems. Adventitious roots grow from the stem just above the anaerobic zone when there is regular flooding (Mitsch and Gosselink 2015). Buttressing bases of trees, also known as stem hypertrophy, are found in several tree species in bottomland hardwood forests. It’s very common to find bald cypress, pond cypress (*Taxodium ascendens* Brongn.), or black gum (*Nyssa sylvatica* Marshall) with buttressed bases. Hypertrophy is an adaptation that helps the tree stabilize in flooded conditions and is caused by ethylene production and by larger cells and lower-density wood, not by aerenchyma. American elm is found to have fluted trunks in bottomlands that help for the same reason (Mitsch and Gosselink 2015). Bald cypress seedlings are adapted to have rapid vertical growth rates so that they can get the photosynthetic organs above water levels when they rise. They also grow a shallow root system with pneumatophores, or "knees", that help gas exchange to the root system and stabilize trees in flooded soils. The pneumatophores usually only form when there is frequent inundation of the soil. Red maple in regularly flooded bottomland areas will have a shallow root system compared to the deep taproots they grow in upland sites (Mitsch and Gosselink 2015).

Native Vegetation Regeneration

Native tree regeneration in bottomland hardwood forests is important for maintaining the species composition of the forest. There are multiple stages of regeneration including seed production and dispersal, germination, emergence, establishment, and survival. If there is prior establishment and survival, sprouting is another successful form of regeneration (Kroschel et al. 2016). Survival of seedlings will depend on internal and external controls, for example, physiological mechanisms or disturbance. Productive regeneration is determined by total seed production; not all seeds will reach the germination stage because of herbivory or other disturbances. The more seeds produced the better chance of regeneration. Seeds are then dispersed by flood, wildlife, or wind. The methods of dispersal for oaks and hickories are wildlife and water, for example, squirrels burying acorns for storage or overcup oaks acorns floating in water during annual flooding. American elm and sycamore are species of bottomland trees that are dispersed by wind because of the winged design of the seeds (Kroschel et al. 2016). Seed germination is controlled by several factors, including soil moisture, light availability, and air temperature. Seed viability is another factor that is important for germination. For example, when overcup oaks seeds float, they can survive better compared to other oaks acorns that can rot in flooded conditions. Early season germination of seeds is at risk because of disturbances like flooding. Once a seedling grows into a sapling, flooding is not as much of an issue as light availability becomes more important. Open areas that are high in light availability and wet provide a better environment for oaks to grow, and dry shaded areas have a better opportunity for competing species, such as sugarberry, to dominate (Kroschel et al. 2016).

Bottomland Hardwood Forests and Disturbance

Following disturbances, like logging and storms, higher elevation areas that are well drained within the forest naturally have pioneer species communities that grow. These stands will break up over time and allow for more shade-tolerant species to grow beneath the pioneer species. Regeneration of more facultative species will follow, like water oak and sweetgum, and these will persist for up to 200 years (Stanturf et al. 2001). Disturbances in permanently flooded areas often do not change the composition of the area. Cypress and gum species grow here and can persist for thousands of years (Hodges 1997; Stahle et al. 2019). Low elevation, poorly drained sites following a disturbance heavily favor black willow as the pioneer species. Black willow populations start breaking up after 30 years of establishment. Vegetation development in this area depends on the rate of sediment deposition. Elm, ash, and facultative wet oaks are the dominant species that regenerate these sites and make up the majority of bottomland forests (Stanturf et al 2001; Hodges 1997).

Major disturbances that impact bottomland hardwood forests in the Gulf Coastal Plain include hurricanes. Hurricanes can alter plant communities and allow invasive species to establish. Conner et al. (2005) found in their study that Chinese tallow invaded a bottomland forest in South Carolina following Hurricane Hugo in 1989. Tallow stem density went from 63 stems/ha before the hurricane to 1269 stems/ha nine years after the hurricane. Another study found that after Hurricane Katrina bottomland forests in Louisiana were affected by delayed mortality of native tree species which opened up habitat for Chinese tallow to invade. Delayed mortality means trees may look like they may have survived the hurricane but could not recover from the damage. The mortality allowed Chinese tallow to become the dominant species in the hurricane effected bottomland hardwood forests (Henkel et al. 2016). Native species of vegetation can also be an issue following hurricanes. One study found that switch cane (*Arundinaria gigantea* (Walter) Muhl.) created dense colonies following hurricane-like damage in a forest and prevented native tree seedlings from growing (Gagnon et al. 2007).

Bottomland Hardwood Forest Management

Managing a bottomland hardwood forest sometimes requires clear-cutting and replanting productive native species. Regeneration of oak species after a clear-cut is successful if there is oak reproduction prior to clear-cut, potential of sprouting from stumps, and all trees in the stand are cut. After clear-cutting, the pioneer species will be fast growing and shade-intolerant species, such as sweetgum. Oaks will outgrow the pioneer species and form the majority of the canopy of the mature stand (Meadows and Stanturf 1997). Managing for cypress and tupelo can be difficult because seedlings are shade intolerant and do not establish in standing water. A light thinning during dry periods will allow sunlight to the forest floor which encourages establishment. Once the seedlings are taller than the depth of floods, a regeneration harvest should be done to establish new seedlings.

Herbicide is a useful tool that can allocate growing space to native species of trees in the bottomlands. One popular method of controlling unwanted woody vegetation in a bottomland hardwood forest is the “hack and squirt” method. This involves cutting into a small tree or shrub and spraying herbicide to the exposed inner part of the stem so that the tree cannot resprout from the stem. This method has been found to be over 90% effective and has minimal effects on non-target trees (Alkire et al. 2012). This method is especially effective for the regeneration of oak seedlings that are shade intolerant. Hack and squirt can remove the overstory, which usually consists of invasive or undesirable species, especially in disturbed areas, and allow the oak seedlings to have all of the proper resources to grow (Lockhart et al. 2000). Compared to basal bark applications, the hack and squirt method gets better translocation of the herbicide to the root system killing the rest of the tree that is left (Webster et al. 2006).

Wetland Mitigation Banks

The Clean Water Act was initially enacted in 1948 as the Federal Water Pollution Control Act, and then later expanded in 1972 when it became known as the Clean Water Act (EPA.gov). Section 404 of the Clean Water Act was passed in 1977 and established a permit program for regulating the discharge of dredged or fill material into Waters of the United States (WOTUS) (Hough and Robertson 2008). Jurisdictional wetlands are included in WOTUS and fall under the protection of section 404. Bottomland hardwood forests are a protected ecosystem under section 404 because they are made up of wetlands and floodplains. In 1977, president Jimmy Carter issued Executive orders 11990 and 11988 which directs federal agencies to "minimize the destruction, loss, or degradation of wetlands and floodplains" (Wiebe and Heimlich 1995).

There are instances when avoidance and/or minimization of impact to a wetland is not possible. As such, the loss of wetland function must be mitigated through the creation and enhancement of wetlands ideally within the same watershed in order to meet the "no net loss of wetlands" goal stated by the Carter administration (Wiebe and Heimlich 1995). The idea and practice of wetland mitigation banking began since this practice provides compensation for "no net loss" of wetland function when unavoidable impacts to wetlands occur. Mitigation banking allows people to buy mitigation credits from mitigation banking companies to compensate for their unavoidable impacts to wetlands. Mitigation banking companies build and manage wetlands to sell credits based on requirements set by the Army Corps of Engineers (Hough and Robertson 2008).

The requirements for a mitigation bank depends on the Army Corps district that the bank is located in. For example, in the Fort Worth District, when planting trees, a minimum density of native approved trees must be met five years after planting, the three most dominant species must be a native species, and not one species may be more than 30% of the surviving trees (USACE 2004). The performance of the wetland must meet standards as well, the wetland must meet the requirements of a jurisdictional wetland, waters must meet the definition of waters of the United States, and buffer and riparian zones function as the intended type of ecosystem component and at the level of ecological performance that was described in the mitigation plan (USACE 2004).

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Chapter II: Long-Term Efficacy of Controlling Chinese Tallow in a Bottomland Hardwood Forest with Herbicide

Introduction

Chinese tallow (*Triadica sebifera* (L.) Small) is a native deciduous tree in eastern Asia and has become a predominant invasive species in southeastern United States forests (Pile et al. 2017). Bottomland hardwood forests are prone to invasion because Chinese tallow has high seed production and can grow under a variety of conditions (Yang et al. 2021). In 2008, 185,000 ha of southern forests were occupied by Chinese tallow (Oswalt 2010). The fourth most common tree in Southeast Texas in 2007 was Chinese tallow, and East Texas had a 174 percent increase of Chinese tallow from 1994 to 2006 (Oswalt 2010). Chinese tallow can spread throughout a forest quickly because a mature tree can produce 310,000 seeds per year and has seed dispersal pathways that include water, wildlife, and human activities (Yang 2019). Seedlings are shade, herbivory, and competition tolerant so Chinese tallow impedes native tree regeneration (Gan et al. 2009).

Chinese tallow, being a facultative species (USDA 2023), grows in a range of areas, like grasslands, bottomland hardwoods, and disturbed areas (Ursbatsch 2018). Because Chinese tallow is a tetraploid plant, it can adapt quickly to changing environments and invade areas quickly (Gao et al. 2016). When flooding occurs, Chinese tallow has adapted to form hypertrophied lenticels and reduce biomass of its roots to help avoid the stress from the lack of oxygen to the point where Chinese tallow has been found to outlast bald cypress (*Taxodium distichum* (L.) Rich.) seedlings (Conner 1994). Conversely, Chinese tallow invasion on sloping landscape is less successful due to limited seed dispersal (Fan et al. 2012). After disturbances, like logging, many species of trees will stump sprout, including Chinese tallow. There are studies that suggest that the size of the tree can determine the likelihood of resprouting. Pond cypress were found to have more sprouts in trees 14 cm dbh and lower than any larger (Randall et al. 2002). Younger and smaller black oak, white oak, and scarlet oak were found to have the highest probability of resprouting (Dey and Jensen 2002).

Bottomland hardwood forests provide a good opportunity for a Chinese tallow invasion because the proper growing conditions and seed dispersal pathways exist (Fan et al. 2012). Bottomland hardwood forests occur on alluvial floodplains along rivers and streams where flooding is a common occurrence (Hodges 1997). Native vegetation in bottomland hardwood forests, including oaks, may not be competitive with Chinese tallow because of its shade and flood tolerance and rapid growth rates (Conner et al. 2005; Fan et al. 2012). Disturbances, like hurricanes, logging, and tornadoes, give Chinese tallow a window to establish and outcompete native regeneration (Conner et al. 2005).

Chinese tallow's range is limited in northern expansion because it does not have a high tolerance of cold climate conditions. No Chinese tallow are found in elevations higher than 162 meters above sea level in the Unites States (Yang et al. 2021). Freeze tolerance for germination and overwinter survivability is most likely determined by genetics. Studies have found that different genes from North Carolina and South Carolina vary on survivability when exposing Chinese tallow to freezing conditions (Park 2009). In February 2021, a major winter storm hit East Texas covering it in snow and ice for a week, hitting a low temperature of -16℃ and a high of -2℃ (Weather.gov 2023). This storm may have impacted Chinese tallow survival in the region, even for large mature trees.

Controlling Chinese tallow in bottomland hardwood forests can be a difficult endeavor. Using herbicides and supplemental planting of native species can give native bottomland hardwoods a competitive advantage after disturbance (Nix 2004). Herbicides, like triclopyr and imazapyr, are commonly used in a foliar, hack and squirt, or basal bark application (Miller et al. 2013). One persistent issue with using herbicide to control Chinese tallow is that failed efforts lead to resprouting and from the roots or the bole (Norman 2020). Other control efforts, as in prescribed fire and mastication, can lead to initial control of Chinese tallow, but long-term control is not effective without frequent retreatment (Pile et al. 2017). Chinese tallow is known as a fire suppresser and mastication can spread seeds and cause stump sprouting (Grace 1998; Donahue et al. 2004). The efficacy of herbicides on long-term control of Chinese tallow requires further study (Enloe et al. 2015). There are a few studies that report on long term control of Chinese tallow over 7 and 8 years (Cutway 2017; Norman 2020). Those that range over

10 years (Oswalt 2010) study the expansion and populations of Chinese tallow over ecosystems. It is important to study the long-term effects of herbicidal control of Chinese tallow.

I hypothesized that stands treated with herbicide would have reduced Chinese tallow density and increased native tree density. This would result in other differences in stand structure, such as basal area, quadratic mean diameter, relative density, stand density index, seedling and sapling density, and arboreal diversity. The number of years since the last herbicide treatment was hypothesized to be correlated to the magnitude of changes in these variables. Additionally, an observational study was conducted on resprouting rates of Chinese tallow believed to be killed during the February 2021 winter storm, to both quantify the incidence of mortality, and to test the hypothesis that smaller top-killed trees would be more likely to resprout than larger trees.

Methods

Study Area

To evaluate the long-term effectiveness of herbicide on controlling Chinese tallow a re-measurement of a paired-plot design (Camarillo et al. 2015) was conducted in the bottomland hardwood forests located at the Pineywoods Mitigation Bank (PMB). Herbicide pretreatment vs. post-treatment impacts on Chinese tallow and native woody species were evaluated to determine effectiveness.

The PMB is a 7,689-hectare wetland and stream mitigation bank in East Texas near Diboll, Texas (31º09’39.5” N, 94º46’18.8” W). Average high temperatures of 35℃ and a low of 22℃ in the summer and a high of 13℃ and a low of 5℃ in the winter are found at the PMB. Average rainfall is around 1520 mm per year (weather.gov 2023).

Known as the largest mitigation bank in Texas, the PMB is located along the Neches River and stretches through Angelina, Polk, and Jasper Counties (Figure 1). The PMB is a mixed hardwood forest with mesic flats that consist of willow oak (*Quercus phellos* L.), water oak (*Quercus nigra* L.), and laurel oak (*Quercus laurifolia* Michx.) cover types in the low-elevation areas of the property, and sweetgum (*Liquidambar styraciflua* L.) and red oak cover types at slightly higher elevations. In the wetlands located on the property, the dominant species are black gum (*Nyssa sylvatica* Marshall) and bald cypress. The extent of Chinese tallow varies in these cover types from patches of only a few square meters up to several hectares (Camarillo et al. 2015).

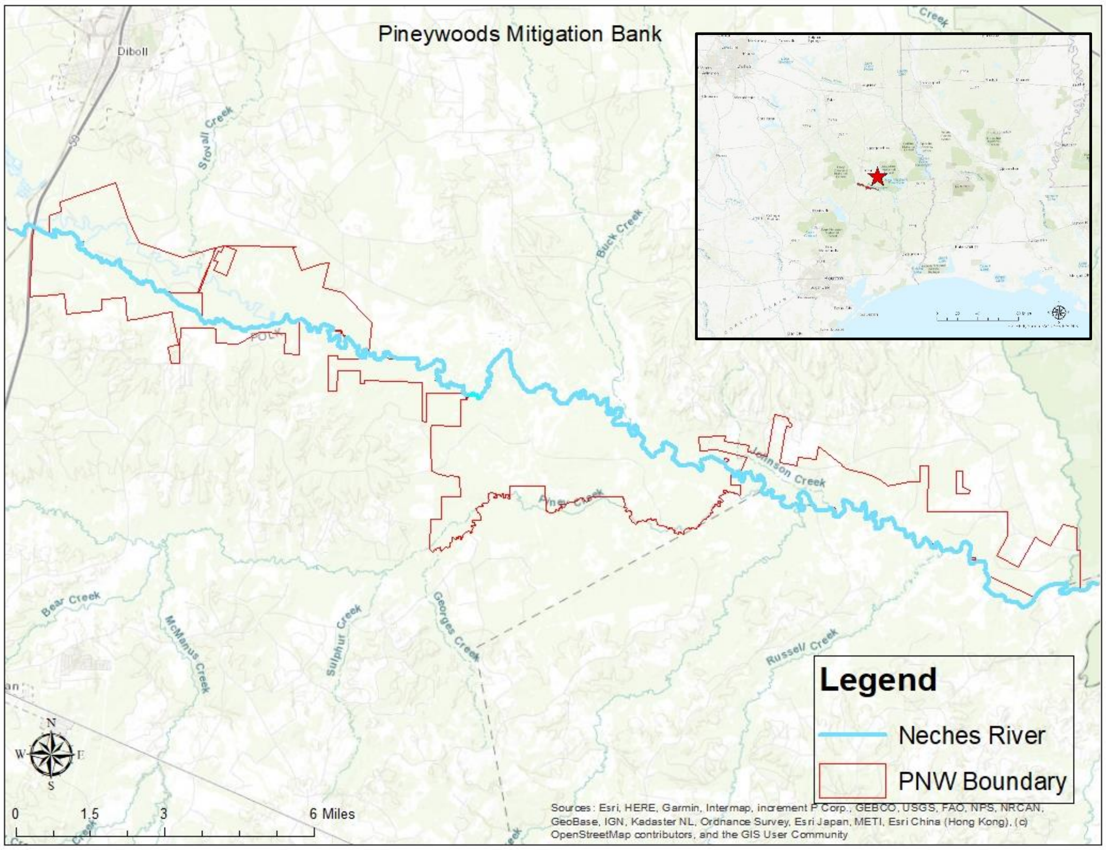

Figure 1. Map of the Pineywoods mitigation bank with a locator map showing location in East Texas.

Operational herbicide treatments were applied to various areas at the PMB from 2011 to 2016 (Figure 2). Chinese tallow trees four inches diameter at breast height (dbh) or larger were injected with imazapyr using a hack-and-squirt method. Anything smaller in diameter was treated with a basal bark application of triclopyr mixed with a surfactant using a backpack sprayer. No further details are available relating to the prescriptions or rates, given their operational nature and the time elapsed since treatment. The treatments are believed to follow guidelines of Miller et al. (2013).

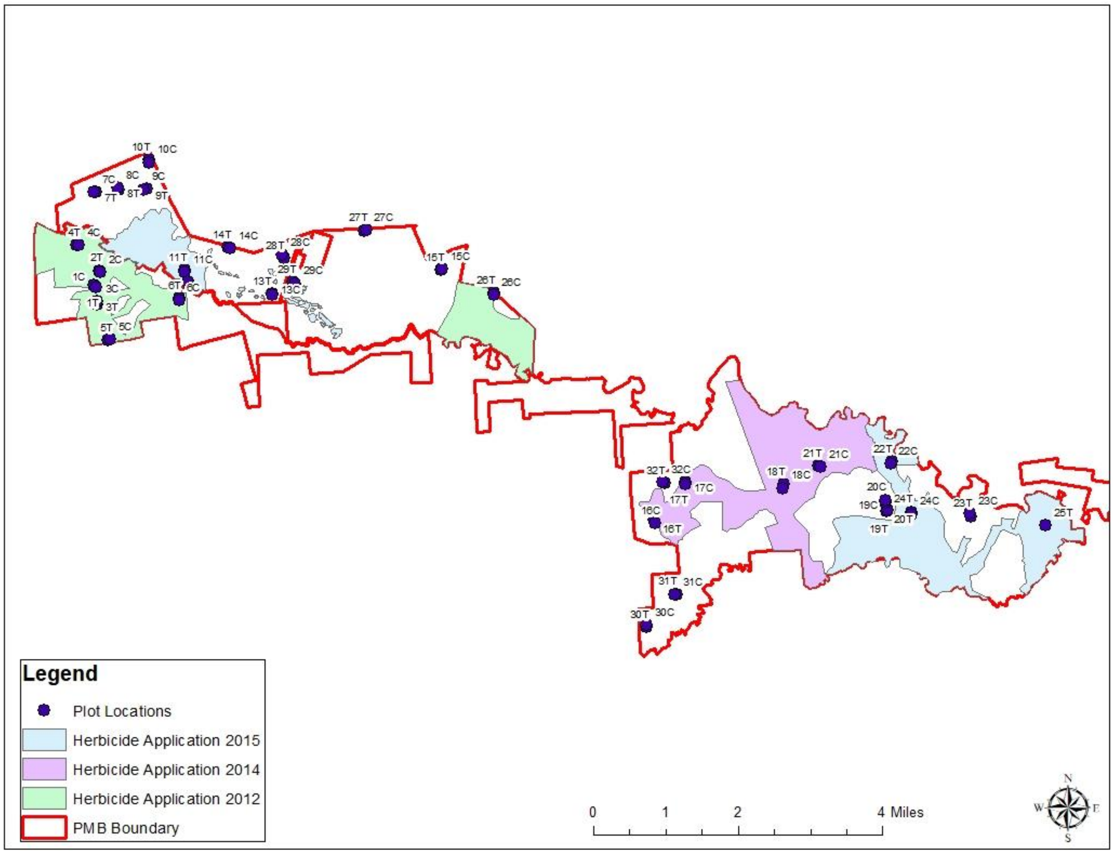

Figure 2. Map of herbicide applications for three different years in the upper and middle sections of the PMB and the plot locations.

Experimental Design and Data Collection

In 2012, throughout the study area, sampling was conducted in a paired plot experimental design (Figure 3). Full details can be found in Camarillo et al. (2015). Briefly, plot locations were initially selected such that one plot, within each pair of plots, had a greater abundance of Chinese tallow (Chinese tallow plot) and the other had little to no Chinese tallow (native plot). Chinese tallow plot and its paired native plot are referred to as plot type. Paired plots were replicated across stands, but not necessarily within each stand. A stand was identified as a contiguous areas with relatively homogenous composition, stand structure, and soils (O'Hara and Nagel 2013). The criteria used to characterize Chinese tallow plots were meeting any of the following thresholds: two Chinese tallow trees per 200 m^2 plot > 25.4 cm dbh; four trees per 200 m^2 plot > 17.8 cm dbh; eight trees per 200 m^2 > 10.2 cm dbh; 16 trees per 50 m^2 plot > 2.5 cm dbh; or 32 trees per 50 m^2 plot < 2.5 cm dbh.

These criteria were determined based on the tallow densities observed at the PMB during the initial study in 2012. The Chinese tallow plots were chosen in 2012 in areas that would be targeted for herbicide application due to their high density of tallow.

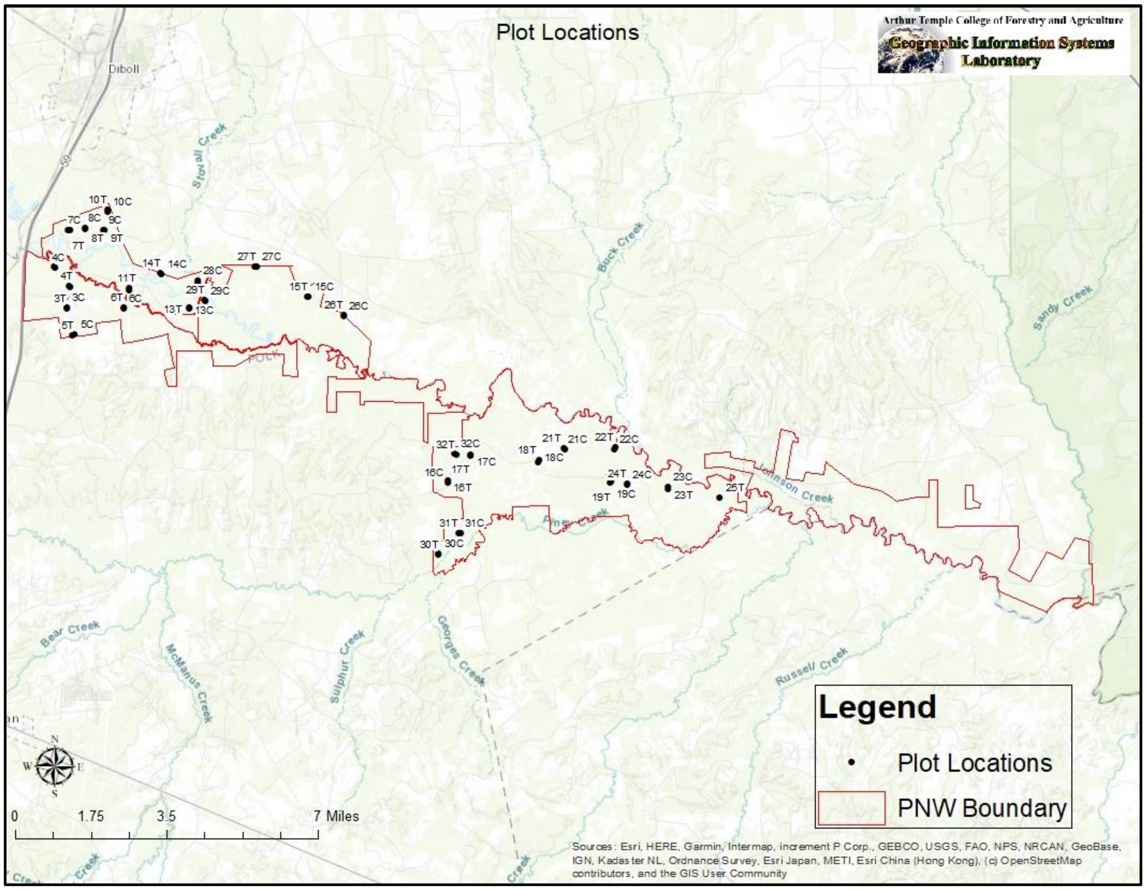

Figure 3. Location of 32 plots on the Pineywoods Mitigation Bank.

The native paired plot was installed in an area that did not meet the criteria for a tallow plot. Native plots were located in an area with a low density of tallow so the area would not likely be targeted for herbicide treatment later. Distance between the Chinese tallow and native plots were randomly generated using Microsoft Excel and ranged from 32 to 96 m at random azimuths (Camarillo et al. 2015). There were 22 paired plots that were installed and measured in 2012 for the purpose of this study. The 22 plots were re-measured in 2022. An additional 10 plots were measured in 2022 for the winter storm observation portion of this study but were excluded from other analyses as they had been treated with herbicide prior to their initial sampling in 2012-13, and thus no pre-treatment data are available.

Each native and Chinese tallow plot included three sets of nested subplots: one overstory (dbh > 10.2 cm), one sapling (dbh 2.5 to 10.2 cm), and four seedling (dbh < 2.5 cm) subplots. The circular overstory subplot was 200 m^2 in area. The circular sapling subplot is 50 m^2. The seedling subplots are located 3 m from the plot center in each cardinal direction and are 1 m^2 in area. In the overstory and sapling subplots, the height and dbh were measured for every tree, and species was recorded. Vines and shrubs were also sampled in the seedling subplot.

Data Analysis

From the data collected, density, basal area (BA), quadratic mean diameter (QMD), stand density index (SDI), and relative density (RD) (using a max SDI of 570 twenty-five cm dbh trees per hectare (Schnur 1937)) were calculated for comparison to the same calculations from the 2012-2013 data. Richness was used along with the Shannon index to ascertain any differences in diversity and evenness between the tallow plots and control plots and their changes over time.

There were 12 plots treated with herbicide between 2012-2022 and 10 plots that were not treated (Table 1). Assumptions were checked by identifying outliers, running a Shapiro-Wilk test to check for normality, and creating a QQ plot to visually check for normality. No outliers were excluded from analysis. A two-way repeated measures analysis of variance (ANOVA) was performed to determine whether there was a significant difference and trend through the years in stand structure metrics between years and herbicide treatment using the Tidyverse, ggpubr, and rstatix packages in R (V 4.3.3). An alpha of 0.05 was used to determine significance. Based on the hypothesis being tested, overstory metrics were divided into two subpopulations; Chinese tallow stems across all plots and only native species of trees across all plots. The subpopulation data was then again split into only tallow plots and both tallow plots and native plots. These subpopulations were then used to test the appropriate hypotheses. Including only Chinese tallow plots give accurate results for Chinese tallow treatment sine there was Chinese tallow in these plots in the original study. For instance, richness, diversity, and evenness were compared with a pairwise T test between year and herbicide treatment. Another example includes stand metrics from only Chinese tallow in tallow plots were compared between year, herbicide treatment, and their interaction using a pairwise T test.

Table 1. Number of plots treated each year.

Year	Plots Treated
2012	4
2014	4
2015	4

A two-way repeated measures ANOVA was used to determine whether there was a significant difference in stand structural metrics between native and tallow plots and the year of measurement. Tukey post-hoc tests were used to determine differences in richness, Shannon's diversity, and evenness between 2022 data and 2012-2013 data. Determining differences in plot type and height class of seedlings per hectare for the 2022 data was done using a two-way ANOVA. A linear regression was visually observed with maximum sprout height and dbh of top-killed Chinese tallow. Years since herbicide treatment was also examined using a linear regression.

Results

Native Plots Compared to Chinese Tallow Plots

All species and plot types were compared between plot type and year. There were no interactions between plot type (native versus tallow) and year for the overstory metrics (Table 2). However, stand density differed between 2012, 2013, and 2022, while basal area differed in the plot type (Table 2). Density was greater in 2012 (mean = 2335.8 trees ha^{-1}) when compared to the other two years (Figure 4). Density was the lowest in 2022 with a mean of 353.3 trees ha^{-1}. Basal area was 104% greater in native plots (Figure 1), but did not vary between the years. Basal area in native plots had a mean of 28.7 m^2 ha^{-1}. QMD and relative density differed in plot type and year, but there was no interaction. In 2022 the QMD was greater (mean = 26.8 cm) than in 2012 and 2013 (Figure 1). QMD was the lowest in 2013 with a mean of 14.34 cm. Native plots had a higher QMD (23.8 cm) than tallow plots (13.7 cm). The year 2012 had the highest mean relative density (110.0%) while 2022 had the lowest mean at 69.9% (Figure 1). Native plots had a higher average (108.7%) than tallow plots (66.0%) (Figure 4).

Table 2. P-values from two-way repeated measure ANOVA for density, basal area, and quadratic mean diameter between native and Chinese tallow plots for 2012, 2013, and 2022. Significant values are bolded.

	Density	Basal Area	QMD	Rel. Density
Plot Type	0.068	**<0.001**	**<0.001**	**<0.001**
Year	**<0.001**	0.062	**<0.001**	**<0.001**
Plot Type * Year	0.054	0.202	0.446	0.308

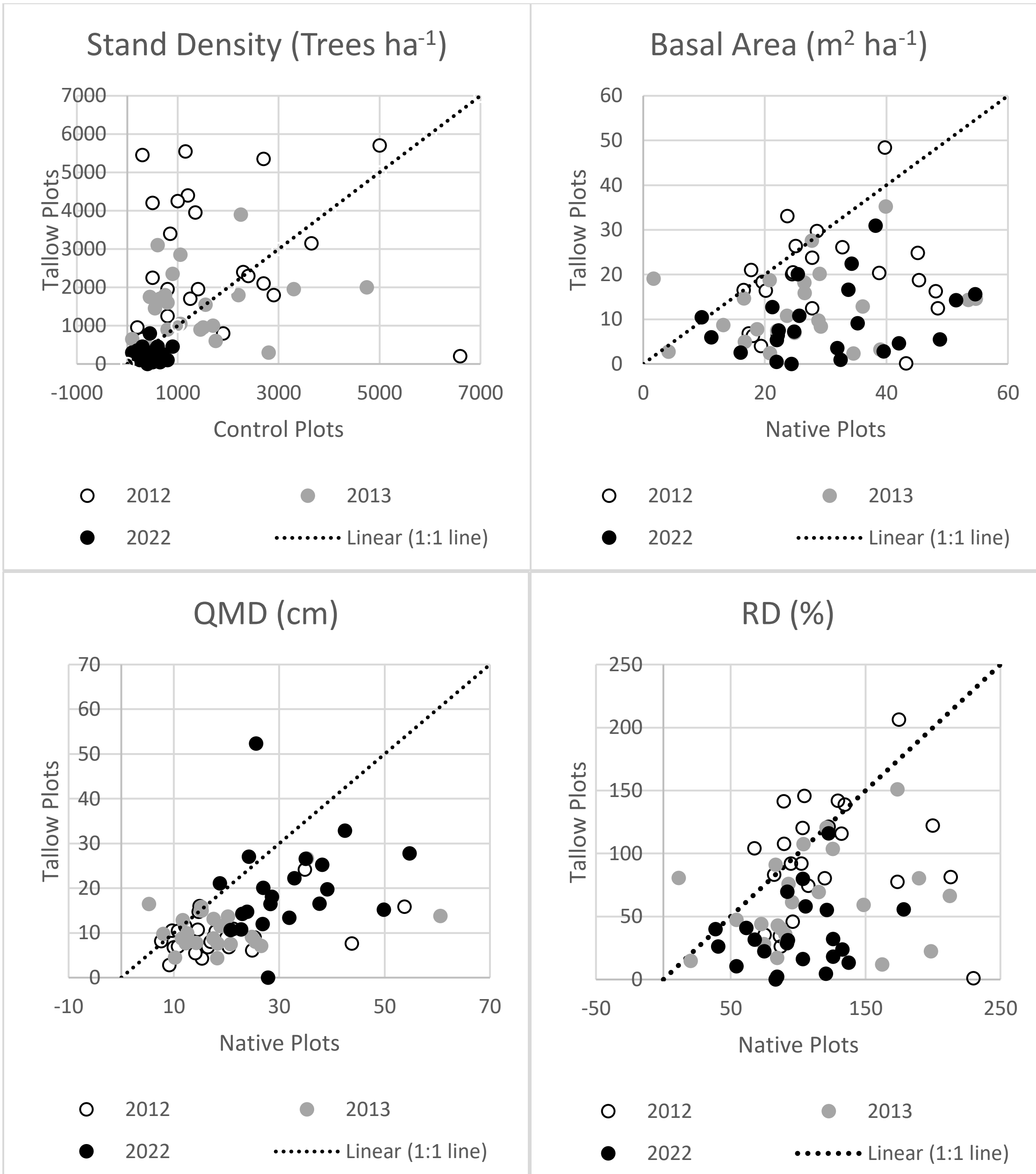

Figure 4. Graphs comparing tallow and native plots for stand density, basal area, and QMD. The dotted line represents a slope of 1 to show which type of plot the majority of the stand metrics are higher in.

Examining 10 years of stand development indicated that the SDI for native and Chinese tallow plots did not exceed an estimated maximum SDI of 1200 trees ha^{-1} at a

QMD of 25.0 cm (Figure 5). Stocking of all species is high after 10 years in both plot types. Native plot SDI was higher than Chinese tallow plots from 2012 to 2022 as indicated by the black lines clustered near the maximum SDI line. Chinese tallow plots (gray lines) showed reduced stocking at a given tree size.

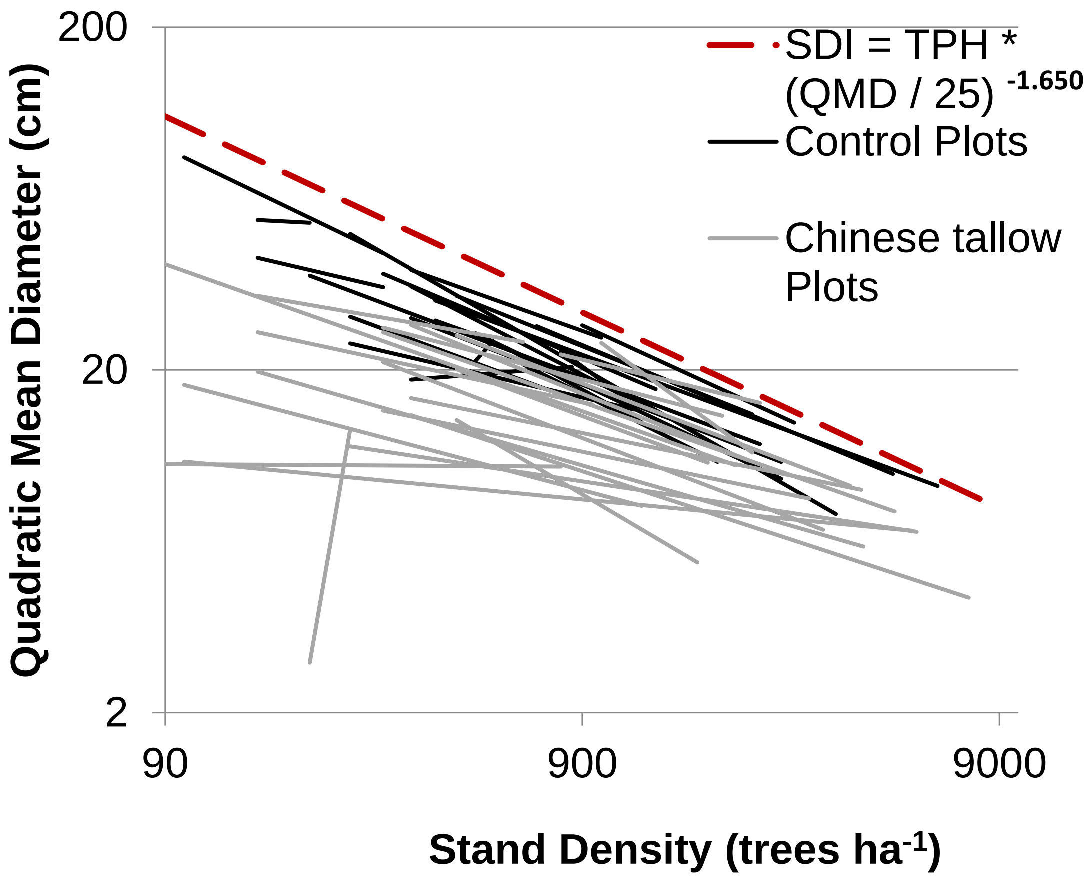

Figure 5. Stand density management diagram for all measured Chinese tallow and native plots in 2012 and 2022. The dashed red line represents an estimated maximum SDI of 1200 trees ha^{-1} at a QMD of 25 cm.

Effect of Herbicide on Stand Structure and Composition

All species of trees across only tallow plots were compared between year and herbicide treatment. Density, basal area, QMD, and relative density differed between the years, but not the herbicide treatment or its interaction with year (Table 3). Density in 2012 was the highest mean (Table 4), while 2022 had the lowest mean (Table 4). Basal area and relative density are similar where 2012 has the highest means (Table 4), and 2022 has the lowest (Table 4). QMD is opposite, where 2022 has the highest mean and 2012 has the lowest (Table 4). No treatment by year interaction was significantly different (Table 3).

Table 3. P-values of repeated measures ANOVA. All species of trees are combined. Only tallow plots are included.

Effect	Density	Basal Area	QMD	Rel. Density
Treatment	0.981	0.077	0.166	0.104
Year	**<0.001**	**0.035**	**<0.001**	**<0.001**
Treatment*Year	0.379	0.476	0.932	0.390

Table 4. Tallow plot only mean and standard error (in parentheses) of density, basal area, and QMD for only Chinese tallow trees, native trees, and all species combined for 2012, 2013, and 2022.

Tallow only	Density	Basal Area	QMD	Rel. Density
2012	2130.4 (369.8)	8.9 (1.2)	9.3 (1.4)	41 (5.2)
2013	976.1 (210.9)	4.1 (0.8)	5.7 (0.9)	22 (4.3)
2022	95.6 (34.7)	1.8 (0.7)	7.4 (1.5)	7.6 (3)
Native only	Density	Basal Area	QMD	Rel. Density
2012	847.8 (207.8)	10.0 (1.9)	13.9 (1.4)	40.8 (8.4)
2013	539.1 (143.6)	10.4 (3.0)	14.5 (1.9)	40.8 (10.3)
2022	204.3 (36.2)	9.21 (1.8)	21.3 (2.4)	31 (6.2)
All spp.	Density	Basal Area	QMD	Rel. Density
2012	2897.7 (369.4)	18.7 (2.3)	9.6 (1.0)	95.7 (10.2)
2013	1534.1 (201.2)	12.3 (1.8)	10.9 (1.0)	60.6 (7.9)
2022	302.2 (50.5)	11.3 (2.3)	20.4 (2.1)	41.7 (8.3)

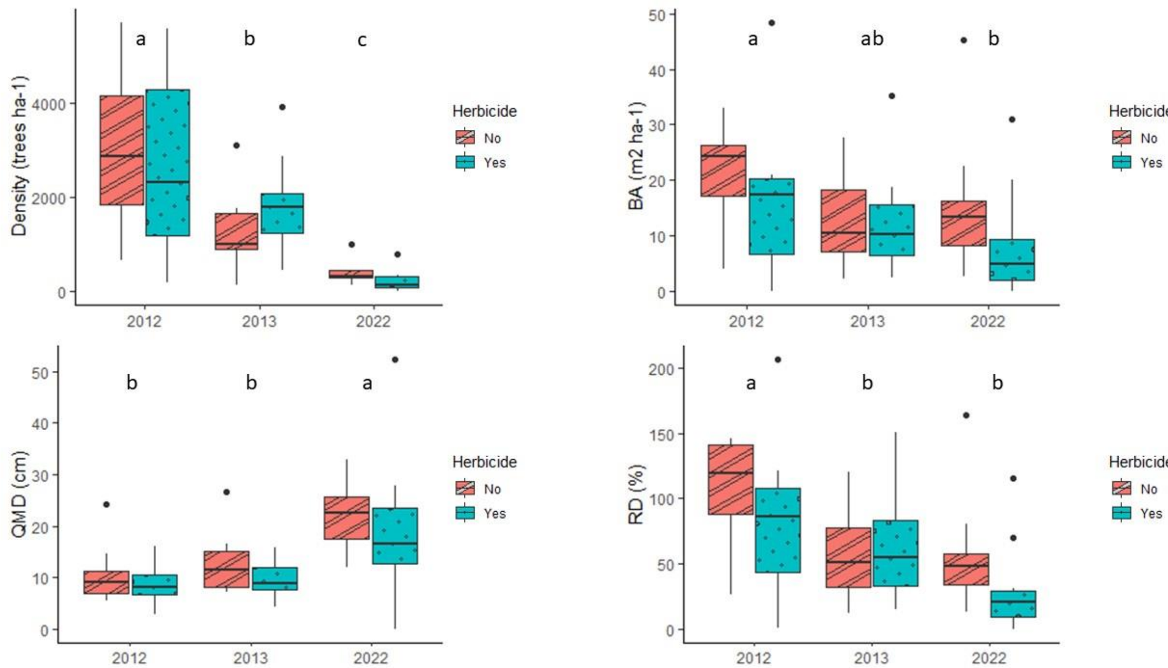

Figure 6. Boxplot showing differences in density, BA, QMD, and RD of herbicide and no herbicide treatments of tallow plots with all tree species included. Letters represent post-hoc comparisons among years only.

Using Chinese tallow tree only data across only tallow plots, density, basal area, and relative density differed between the years (Table 5). Relative density also differed in the treatment by year interaction (Table 5). There was a large difference in Chinese tallow density between 2012 and 2022 (Table 4). Basal area and relative density were also highest in 2012 and lowest in 2022 (Table 4).

Table 5. P-values of repeated measures ANOVA. Only Chinese tallow tree are included from only tallow plots.

Effect	Density	Basal Area	QMD	Rel. Density
Treatment	0.588	0.558	0.285	0.478
Year	**<0.001**	**<0.001**	0.137	**<0.001**
Treatment*Year	0.414	0.084	0.129	**0.040**

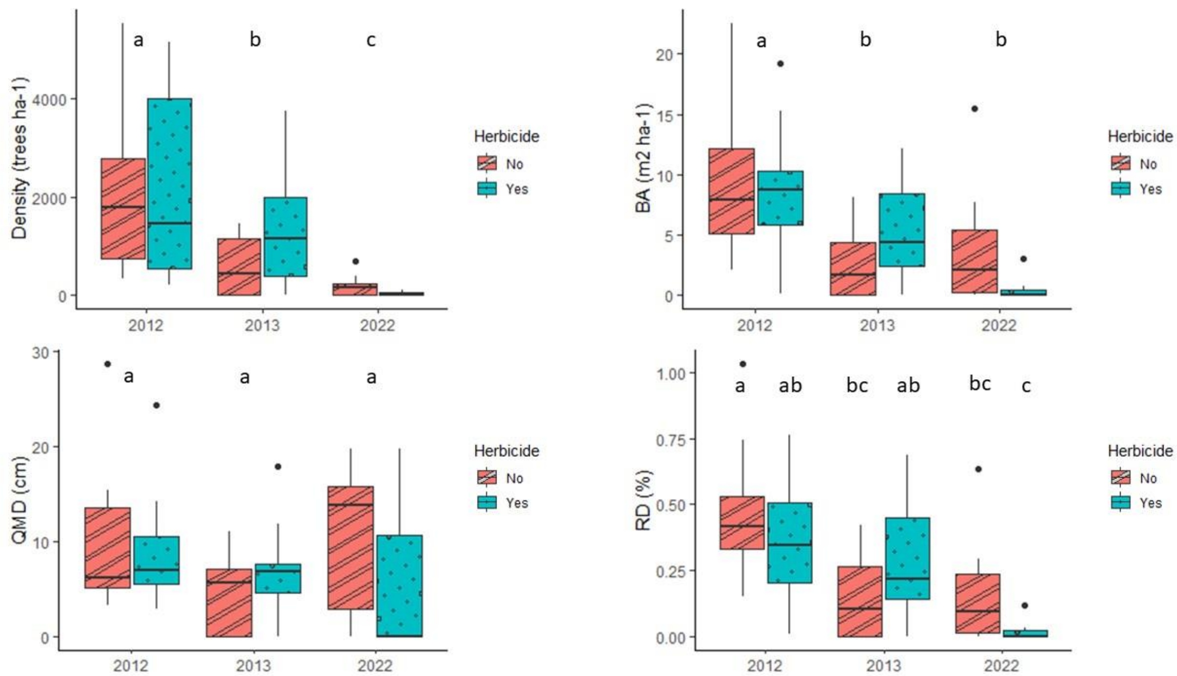

Figure 7. Boxplot showing differences in density, BA, QMD, and RD of herbicide and no herbicide treatments of tallow plots with only Chinese tallow included. Letters represent post-hoc comparisons of years only, except for RD which shows the year x treatment interaction.

Density and QMD differed between the years when running the ANOVA on native species only across only tallow plots (Table 6). QMD also differed between the treatment types (Table 6). Naturally over time, the trees decreased in density (Table 4) due to self-thinning allowing the remaining trees to grow shown by the increase in QMD (Table 4). Plots that were not treated with herbicide had a higher mean QMD (19.4 cm) than plots that were treated with herbicide (14.3 cm). No treatment by year interaction was significantly different (Table 6).

Table 6. P-values of repeated measures ANOVA. Only Native species included from only tallow plots.

Effect	Density	Basal Area	QMD	Rel. Density
Treatment	0.527	0.059	**0.026**	0.063
Year	**0.014**	0.920	**0.011**	0.647
Treatment*Year	0.883	0.843	0.851	0.901

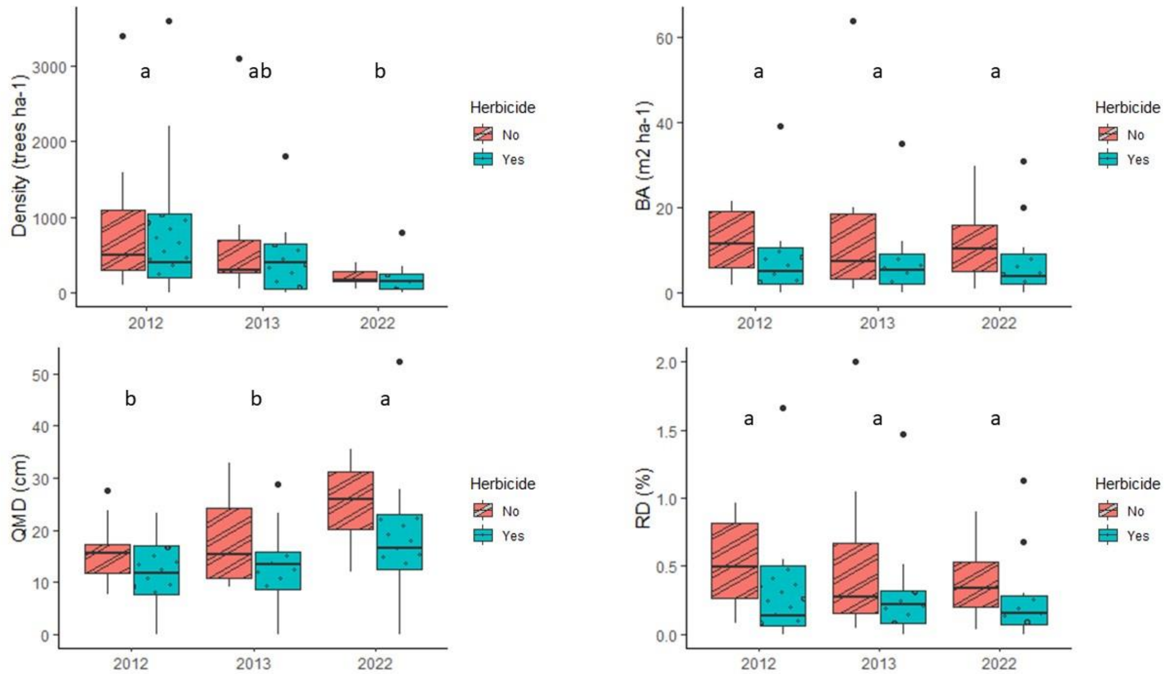

Figure 8. Boxplot showing differences in density, BA, QMD, and RD of herbicide and no herbicide treatments of tallow plots with only native species included. Letters represent post-hoc comparisons among years only.

Regressions showed no difference in years since treatment for any of the stand metrics (Table 7). Visually, all of the herbicide treated plots showed similar trends (Figure 9). Density standard error for overstory and sapling trees is suggests high variability (Table 7). Mean density for sapling trees is slightly greater 10 years since treatment than 6 years (6 years mean = 1200 trees ha^{-1}, 10 years mean = 2000 trees ha $^{-1}$).

Table 7. P-values and R^2 values for simple linear regressions between years since treatment and stand structure metrics for Chinese tallow plots treated with herbicide in the 2022 data.

	Density	BA	QMD	SDI
Overstory P-value	0.797	0.335	0.544	0.272
Overstory T-value	-0.261	-0.991	0.619	-1.134
Overstory Standard Error	39.96	1.266	1.862	25.24
Overstory R^2	0.003	0.051	0.020	0.066
Sapling P-value	0.864	0.061	0.106	0.100
Sapling T-value	0.174	1.994	1.702	1.733
Sapling Standard Error	296.84	0.572	0.5885	18.76
Sapling R^2	0.001	0.180	0.138	0.143

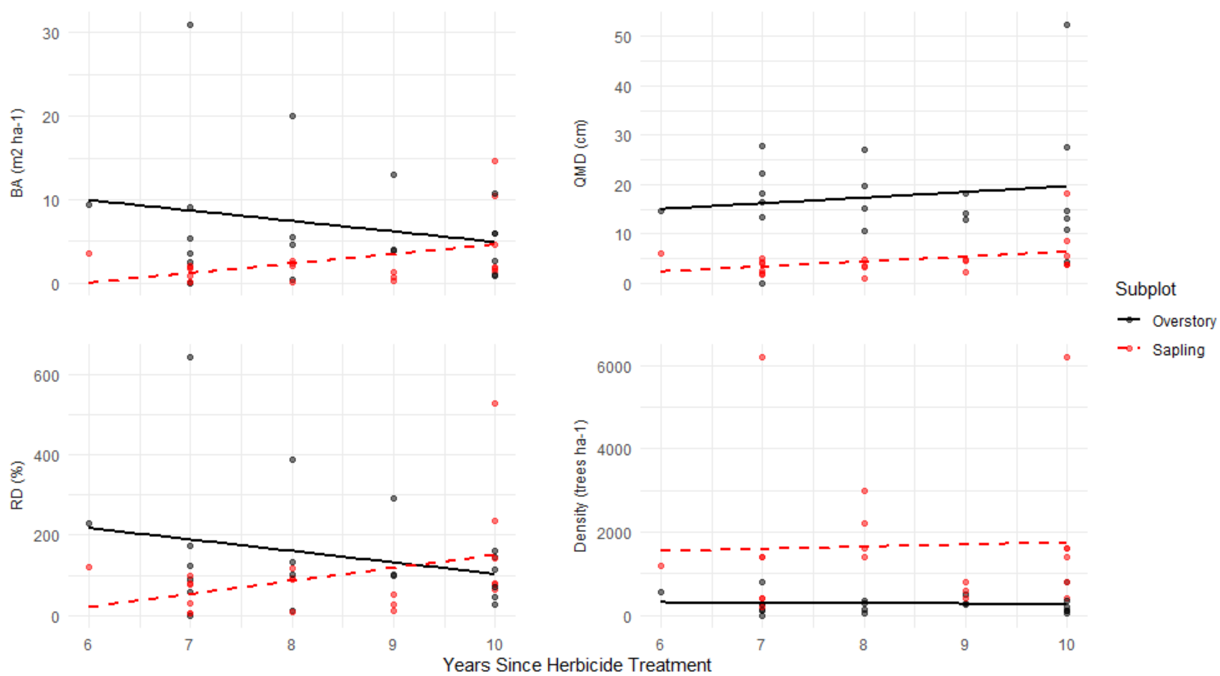

Figure 9. Years since treatment regression with best fit lines for overstory and sapling data.

Diversity

All overstory tree species across all plot types were used to compare between years and herbicide treatment. Overstory richness differed from 2022 to 2012 and 2013 in all types of plots (Table 8). Mean richness for all plots in 2022 was 3.07 and 2012 had the highest mean 4.13. Diversity in the overstory did not differ between the years for any of the plots (Table 8). In all and native overstory there was a difference between 2022 and the 2012 and 2013 data (Table 9). The highest mean diversity for all plots was in 2013 at 0.70 and the mean for 2022 was 0.42. There was a difference in overstory tallow plot evenness between 2022 data and 2012 data (Table 8). Mean overstory evenness for 2012 was 0.55 and 2022 was 0.41. Richness differed between herbicide treated and non-treated plots (Table 8). Herbicide treated plots had a mean richness of 4.54, while non-treated plots had a mean of 5.63. Diversity also differed between the treatment types (Table 8). Mean diversity in herbicide treated plots was 0.75, and mean diversity in non-treated plots was 0.95.

Table 8. P values of repeated measures ANOVA for overstory Richness, Diversity, and Evenness.

Effect	Richness (Species)	H	E
Herb	**0.040**	**0.009**	0.070
Year	**<0.001**	0.766	**<0.001**
Herb* Year	0.758	0.668	0.932

Table 9. P-values from Tukey post-hoc overstory comparisons for 22 plots of three different years of data collection. H is Shannon's diversity and E is evenness. Significant values are bolded.

	Year	Richness (species)	H	E
Native Plots	2013-2012	0.999	0.901	0.815
	2022-2012	**<0.001**	0.994	**0.002**
	2022-2013	**<0.001**	0.940	**0.014**
Tallow Plots	2013-2012	0.738	0.999	0.127
	2022-2012	**0.008**	0.729	**<0.001**
	2022-2013	**0.059**	0.720	0.140
All Plots	2013-2012	0.850	0.947	0.152
	2022-2012	**<0.001**	0.890	**<0.001**
	2022-2013	**<0.001**	0.721	**0.003**

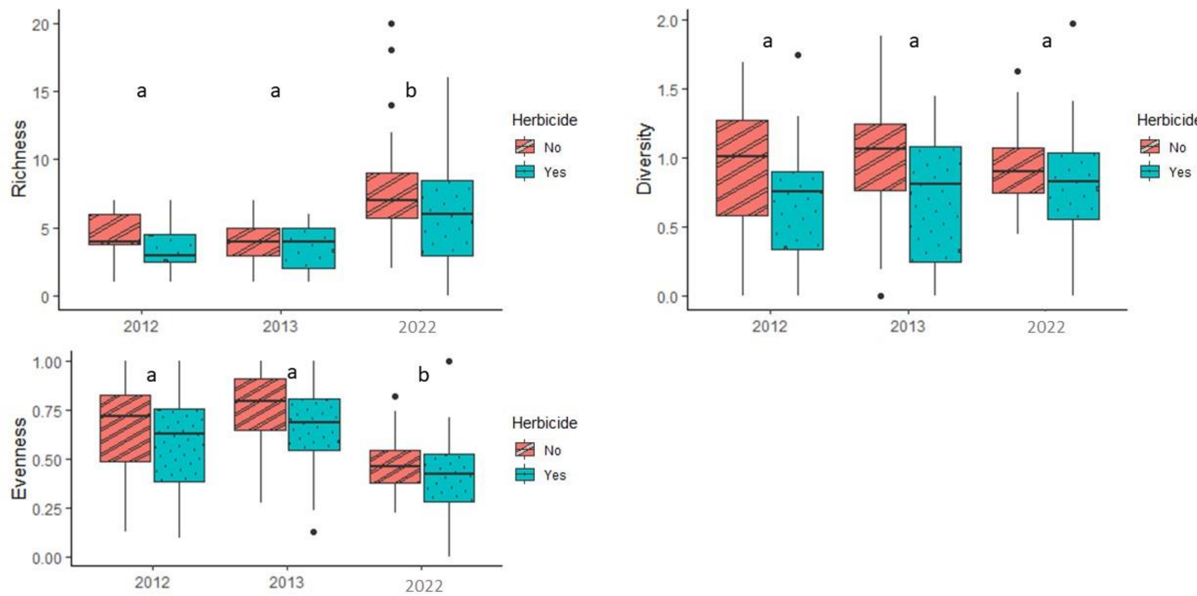

Figure 10. Pairwise T test box plot for overstory Richness, Diversity, and Evenness.

All sapling tree species across all the types of plots were used to compare between years and herbicide treatment. Sapling richness, diversity, and evenness did not differ from the interaction (Table 10). Sapling diversity and evenness was different between 2022, 2012, and 2013 (Table 10). Mean diversity for all sapling plots in 2012 was 1.56 and 2022 was 0.78. Mean evenness for all sapling plots in 2012 was 0.79 and 2022 was 0.44.

Table 10. P values of repeated measures ANOVA for sapling Richness, Diversity, and Evenness.

Effect	Richness	H	E
Treatment	0.098	0.611	0.961
Year	0.736	**<0.001**	**<0.001**
Treatment*Year	0.926	0.567	0.654

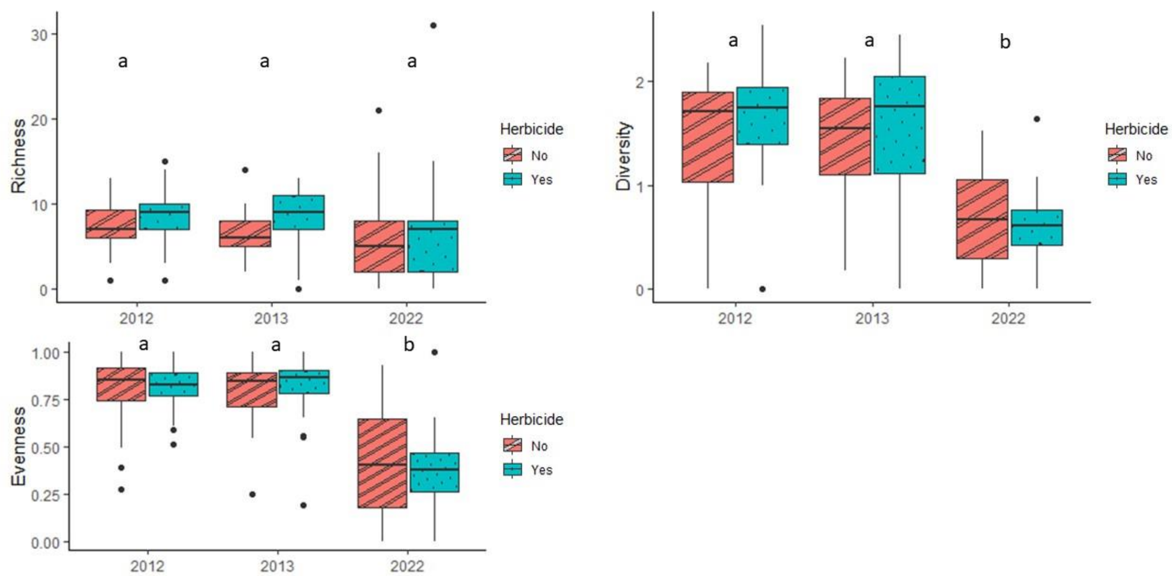

Figure 11. Pairwise T test box plot for sapling Richness, Diversity, and Evenness.

Table 11. P-values from Tukey post-hoc sapling comparisons for 22 plots of three different years of data collection. H is Shannon's diversity and E is evenness. Significant values are bolded.

	Year	Richness	H	E
Native Plots	2013-2012	0.986	0.959	0.998
	2022-2012	**0.097**	**<0.001**	**<0.001**
	2022-2013	**0.069**	**<0.001**	**<0.001**
Tallow Plots	2013-2012	0.714	0.684	0.997
	2022-2012	0.166	**<0.001**	**<0.001**
	2022-2013	0.546	**<0.001**	**<0.001**
All Plots	2013-2012	0.779	0.706	0.999
	2022-2012	0.999	**<0.001**	**<0.001**
	2022-2013	0.802	**<0.001**	**<0.001**

<u>Seedling</u>

Table 12. Mean sapling diversity for seedling data. H is Shannon's diversity and E is evenness for data collected in 2022.

	Richness	H	E
Mean Tallow	3.937500	1.088701	0.531920
Mean Control	3.838710	1.019133	0.497930
Mean Total	3.888889	1.054469	0.515195

Table 13. Mean sapling diversity for seedling data excluding vines and shrubs.

	Richness	H	E
Mean Tallow	3.437500	0.894137	0.715388
Mean Control	3.516129	0.863719	0.695599
Mean Total	3.476190	0.879170	0.705650

Table 14. Two-way ANOVA results for seedlings per hectare with a plot type and height class interaction for the 2022 data. Significant values are bolded.

	Seedlings per Hectare
Plot Type	0.3142
Height Class	**0.0258**
Plot Type * Height Class	0.8456

Table 15. Two-way ANOVA results for seedlings per hectare with a plot type and height class interaction for 2022 data excluding vines and shrubs.

	Seedlings per Hectare
Plot type	0.884
Height Class	0.143
Plot type * Height Class	0.814

Seedlings per hectare based on plot type and height class interaction was not different (Table 13) while the height class was different. There are more seedlings per hectare in height class < 0.30 m (37653.84 seedlings ha^{-1}) than the other two height classes. Richness, Shannon's diversity, and evenness were slightly higher in tallow plots than in control plots, and tallow plots were higher than the overall average (Table 12).

February 2021 Winter Storm Impact on Mortality and Sprouting

A total of 605 Chinese tallow was sampled across all plots and a total of 75 top-killed Chinese tallow were observed, showing an 87.6% survival rate after 16 months (Table 15). Only one Chinese tallow that was top-killed did not produce sprouts. Sprout height and dbh of top-killed tree correlations were not observed (Figure 12). Histograms show the diameter and height distributions of Chinese tallow across the site and show most Chinese tallow are sapling size.

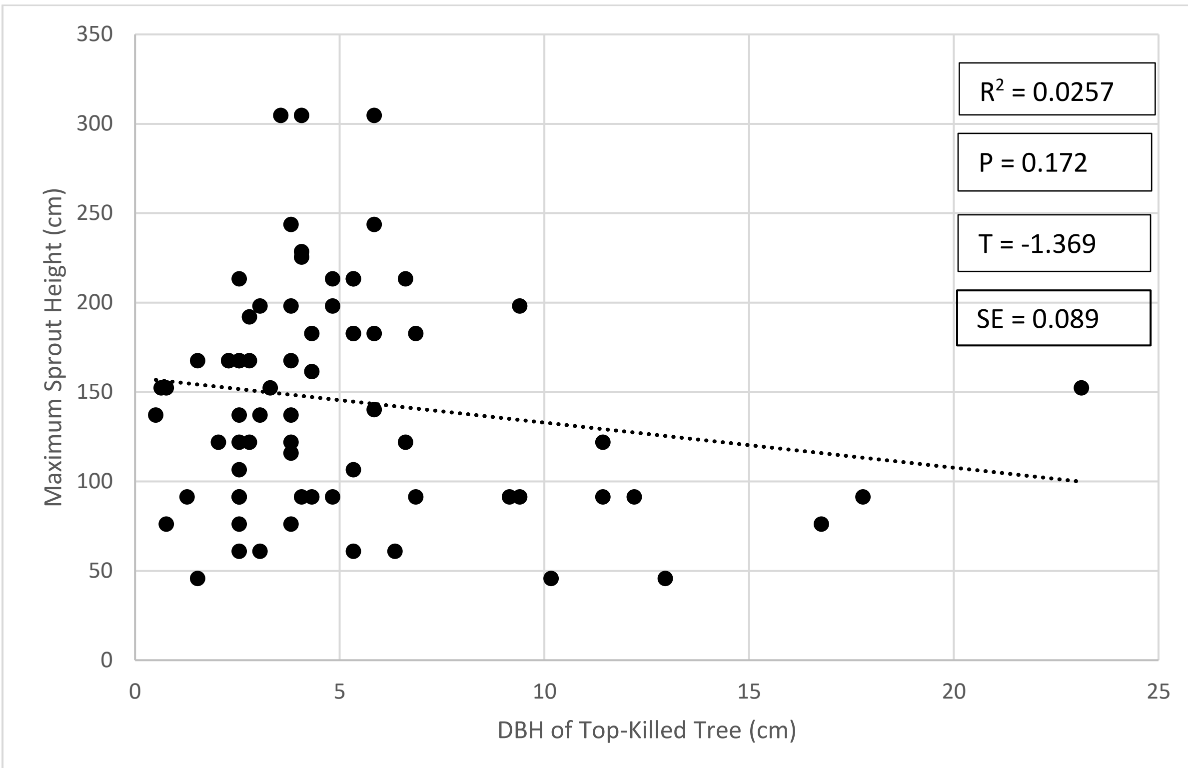

Figure 12. Linear regression of top-killed Chinese tallow seedling sprout height compared to the DBH of the tree.

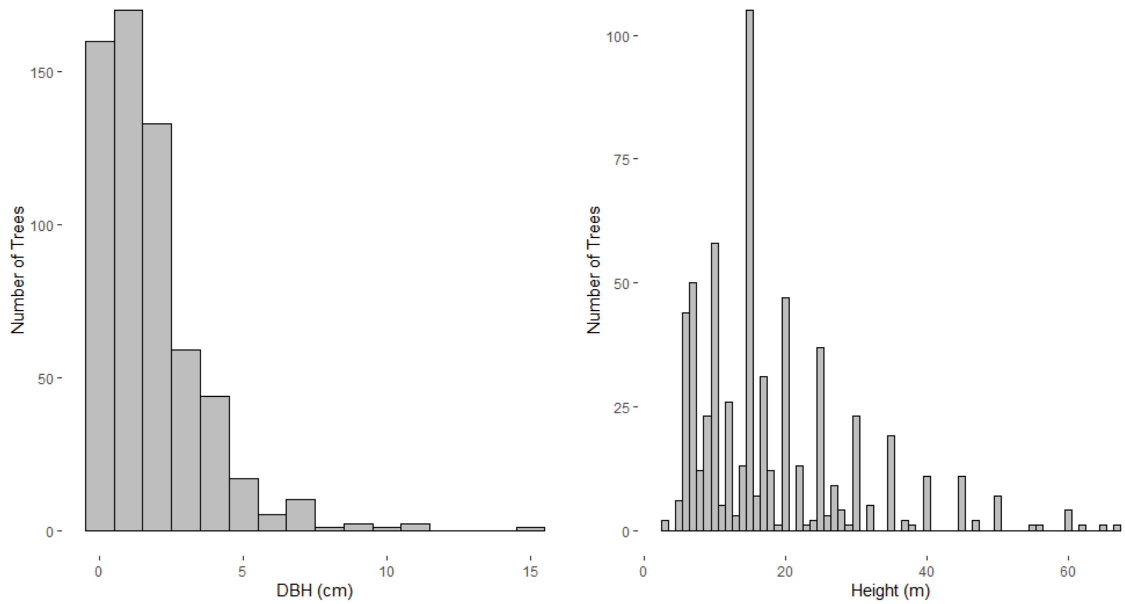

Figure 13. Histogram of dbh and height distribution of Chinese tallow recorded in the Pineywoods Mitigation Bank.

Discussion

Invasive species can alter ecosystem dynamics (i.e., biodiversity, hydrological and soil processes) in bottomland hardwood forests, which in turn, has consequences for the ecosystem services these systems provide (Catford 2014; Huddle et al. 2011; Levine et al. 2006; Raheem et al. 2024; Weng et al. 2012; Yang et al. 2021; Zou et al 2006) Land managers use management techniques, like herbicide, to minimize the effect of invasive species and improve native populations so that the ecosystem can function correctly. Analyses did not support my hypotheses that Chinese tallow and native tree densities responded to herbicide application, whether assessed by trees per hectare, basal area, or quadratic mean diameter. However, I observed relative density was lower in Chinese tallow plots treated with herbicide. I also found that the number of years since the last herbicide treatment did not correlate to the magnitude of changes in Chinese tallow and native tree densities. The observational study on Chinese tallow impacted by the 2021 winter storm primarily showed that trees initially top-killed almost all resprouted. However, the correlation between maximum seedling sprout height relating to tree dbh was weak and non-significant. Control of Chinese tallow has historically been hard to predict, as there are numerous different ecological contexts and processes that may lead to varied responses of Chinese tallow in bottomland hardwoods.

Chinese Tallow Control

Previous studies have reported on a number of management techniques to control Chinese Tallow (Cheng et al. 2021; Cutway 2017; Enloe et al. 2015; Gresham 2020; Norman 2020; Pile et al. 2017). Cutway (2017) found that hand removal of Chinese tallow had no effect on Chinese tallow density in an 8-year study in riverine woodlands, mainly due to vigorous sprouting of removed individuals. Our study found that removal of Chinese tallow using herbicide didn't result in desired mortality allowing populations to remain. Using fire as a control method has been found to increase invasive risk of Chinese tallow by promoting seedling germination under open and closed canopy covers. Studies suggest that fire will only be effective if Chinese tallow seed sources are removed as well (Cheng et al. 2021; Fan et al. 2021). Short-term studies investigating herbicide effects on Chinese tallow density have shown that herbicide applications may lead to reductions; however, these effects may be temporary with long-term studies showing that Chinese tallow densities may be able to recover from initial herbicide applications (Bruce et al. 1997; Enloe et al. 2015; Gresham 2010). Because we don't have any data recorded shortly after the herbicide treatments, we do not know what the initial response to the application was. Regardless, several years after treatments Chinese tallow remains prevalent in the PMB. Pile et al. (2017) utilized a combination of herbicide, prescribed fire, and mastication to reduce Chinese Tallow density in bottomland forests of South Carolina. They found that this combination of management initially reduced Chinese tallow density and remained low throughout the study but did not eradicate Chinese tallow (Norman 2020). They suggested that the additional levels of control (prescribed fire and mastication) did not provide additional levels of control and are unnecessary costs that land managers do not need to allocate to control methods.

Chinese tallow control methods are costly and land managers need to find not only a cost-efficient method, but an effective method in maintaining an ecosystem to their needs (Gao 2016; Norman 2020). Using all three types of control methods is hard for land managers to justify, so herbicide is the most widely used management strategy for control once Chinese tallow establishment has occurred. Therefore, investigating the effects of herbicide across different spatiotemporal scales is vital to recognizing when, where, and how herbicides should be applied. The most Chinese tallow infested forest type is oak/gum/cypress and elm/ash/cottonwood which is the common cover type found in the PMB (Nepal et al. 2021). This study shows how prevalent Chinese tallow is in an East Texas bottomland hardwood forest and how an herbicide treatment did not fully control Chinese tallow.

Studies that specifically focus on herbicide as a management of control, have shown that different herbicides have varying levels of success. Triclopyr is a common herbicide used in treatments, and the herbicide used in our study. In one study, triclopyr did not produce the best results and led to treated Chinese tallow resprouting (Enloe et al. 2015). However, aminocyclopyrachlor and fluroxypyr were found more effective than triclopyr (Enloe et al. 2015). Like triclopyr, aminocyclopyrachlor and fluroxypyr mimic auxin, killing the plant by causing an overdose of auxin (Minogue et al. 2011; Guo et al. 2019). This study used three different treatment applications, like cut stump, basal bark, and foliar applications with all three having similar results. Our method of application in this study was similar, using hack-and-squirt and basal bark methods.

At the PMB, change in Chinese tallow density was not correlated with years since treatment. Similar studies have found that change in Chinese tallow density was correlated with the year (Norman 2020), but not correlation was found with the treatment. Overall tree density was decreased significantly throughout the years but was not correlated with herbicide treatment. Relative density of Chinese tallow in the Chinese tallow plots did decrease significantly over the ten years and was found to be correlated with herbicide treatment. The original study that was the foundation of this study (Camarillo et al. 2015) found that Chinese tallow and native trees shared an inverse stocking relationship and areas with greater Chinese tallow densities had reduced overstory diversity. Our diversity was found to be lower than it was 10 years ago where Chinese tallow was still prevalent.

The herbicide treatments on the PMB did appear to be operationally successful despite the lack of statistical significance. The low sample size of this study causes high variability and can cause the means to not be significant even when it looks like the treatments did work. There are virtually no Chinese tallow overstory trees left in herbicide treated Chinese tallow plots when visually looking at results.

<u>Stand Density Index</u>

After 10 years we had found that the stands did not reach our estimated maximum SDI and had a higher maximum SDI than what is reported for mixed hardwood species from the central hardwoods region, which was between 530 and 620 (Gingrich 1967; Williams 2003; Johnson et al. 2009). These updated 10-year maximum SDI recommendations match the one-year results originally reported in (Camarillo et al. 2015). It appears that SDI in bottomland hardwoods is substantially greater than the upland hardwood stands previously reported in the central hardwoods region (Schnur 1937).

<u>Diversity</u>

Biodiversity is important for a bottomland hardwood ecosystem to thrive and an invasive species can alter community composition by decreasing overall diversity or decreasing evenness leading to the invasive species to be more dominant (Zelder and Kercher 2004). Our results suggested that after an herbicide treatment, stand diversity metrics did not improve. The lower numbers are likely caused by Chinese tallow remaining and displacing native tree populations to less competitive areas by monopolizing the resources and becoming more dominant (Rice and Pfennig 2008). Entire community composition changes happen when an invasive species negatively impacts more than one native species, leading to a decrease in biodiversity (Lockwood et al. 2013), which is what we are finding at the PMB. Similarly, other invasive species, like *Ligustrum sinense* Lour., have been found to reduce biodiversity once established in an ecosystem (Hart and Holmes 2013; Hejda et al. 2009; Zedler and Kercher 2004).

Invasion characteristics by Chinese tallow can largely differ because of a number of factors. It is recommended that management decisions of invasive species need to consider the effect on community characteristics determined by the character of the invaded community, especially the dominance in a stand of the dominant native compared to the dominant invader (Hejda et al. 2009). As well as examining the traits of species communities forming once an invasive species is established rather than reduction of stand diversity metrics. Other studies in bottomland hardwood forests have found that differences in anthropogenic actions on sites lead to varying levels of Chinese tallow invasion based on the tree's characteristics (Neyland and Meyer 1997). Following the recommendations from Hejda et al. (2009), our results provide a further understanding of Chinese tallow's impact to a bottomland hardwood forest community post herbicide treatment.

Winter Storm Observational Study

The observational study on top killed Chinese tallow from the 2021 winter storm showed no correlation between maximum seedling sprout height relating to tree dbh. Almost all of the top-killed Chinese tallow had sprouted seedlings from the tree. Other studies found numerous Chinese tallow resprouting after being top-killed by fire and stated that prescribed fires in the growing season can help reduce the carbohydrates needed for resprouting (Pile et al. 2017). Freezes do not have the same effect as prescribed fire and do not happen during the growing season, therefore the top-killed Chinese tallow has the ability to aggressively resprout from the live stump. The winter storm could explain why tree density was lower overall in 2022, but Chinese tallow was the only species observed to have snags remain and sprouts growing out of the snag.

Conclusion

We found that the herbicide treatment was not correlated with changes in Chinese tallow and native tree density, but was correlated to the decrease in relative density of Chinese tallow. Similarly, number of years since herbicide treatment was not correlated with the changes in Chinese tallow and native tree density. Future studies should focus on using different types of herbicides, multiple-year treatments, and possibly supplemental planting to control the regeneration of native trees. Current and future studies controlling Chinese tallow are important for land managers in East Texas and the rest of the Southeast to minimize Chinese tallows ecological impact on bottomland hardwood forests. Additionally, we found the possibility that a freeze can top-kill mature Chinese tallow and are likely to resprout afterwards.

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Appendix

Table 1. Trees per hectare (trees ha^{-1}), basal area (m^2ha^{-1}), quadratic mean diameter (cm), and relative density (%) of overstory and sapling trees in the native and Chinese tallow Plots collected in 2022.

Plot	Plot Type	Subplot	TPH Trees ha^{-1}	BA m^2ha^{-1}	QMD cm	RD %
1	Tallow	Overstory	1400	4.51223	6.405988	26
1	Tallow	Sapling	3200	10.45135	6.448604	62
1	Control	Overstory	400	28.13443	29.92565	91
1	Control	Sapling	400	3.552019	10.63316	17
2	Tallow	Overstory	550	11.38749	16.23631	47
2	Tallow	Sapling	1400	1.509988	3.70576	11
2	Control	Overstory	600	25.49499	23.25982	91
2	Control	Sapling	600	2.3947	7.128603	13
3	Tallow	Overstory	1950	9.745758	7.977101	53
3	Tallow	Sapling	6400	16.34841	5.702986	102
3	Control	Overstory	750	29.48429	22.37275	107
3	Control	Sapling	400	13.28384	20.563	49
4	Tallow	Overstory	650	7.138495	11.825	33
4	Tallow	Sapling	1600	1.894073	3.882334	13
4	Control	Overstory	300	18.40032	27.94516	61
4	Control	Sapling	1000	18.25464	15.24548	77
5	Tallow	Overstory	1250	7.853713	8.944108	41
5	Tallow	Sapling	1600	46.04856	19.14267	178
5	Control	Overstory	2000	12.42067	8.892257	65
5	Control	Sapling	3000	15.0259	7.985723	82
6	Tallow	Overstory	1450	38.55791	18.4004	151
6	Tallow	Sapling	1000	6.439239	9.054653	33
6	Control	Overstory	1150	20.61919	15.1092	87
6	Control	Sapling	200	8.954535	23.87597	31
7	Tallow	Overstory	1150	22.40508	15.74994	93
7	Tallow	Sapling	2200	9.751079	7.512242	54
7	Control	Overstory	400	34.56632	33.17043	107
7	Control	Sapling	200	2.846683	13.46198	12

Table 1.
Continued

Plot	Plot Type	Subplot	TPH Trees ha^{-1}	BA m^2ha^{-1}	QMD cm	RD %
8	Tallow	Overstory	1400	14.39683	11.44258	68
8	Tallow	Sapling	1600	2.250035	4.231449	15
8	Control	Overstory	250	21.45121	33.05296	66
8	Control	Sapling	0	0	0	0
9	Tallow	Overstory	1600	6.444559	7.161288	36
9	Tallow	Sapling	4200	11.9887	6.028593	73
9	Control	Overstory	700	56.38844	32.02584	178
9	Control	Sapling	1000	11.98465	12.35285	55
10	Tallow	Overstory	550	17.04488	19.86416	65
10	Tallow	Sapling	1000	2.012642	5.062183	13
10	Control	Overstory	550	32.47488	27.41871	109
10	Control	Sapling	600	9.774387	14.40203	42
11	Tallow	Overstory	950	4.695912	7.933279	25
11	Tallow	Sapling	1600	3.654374	5.392636	23
11	Control	Overstory	1500	33.61801	16.89254	136
11	Control	Sapling	1400	3.708085	5.807181	22
12	Tallow	Overstory	900	27.94162	19.88196	106
12	Tallow	Sapling	1000	16.30281	14.4074	70
12	Control	Overstory	550	37.97089	29.64822	123
12	Control	Sapling	1200	4.616105	6.998449	26
13	Tallow	Overstory	1600	10.27273	9.04144	53
13	Tallow	Sapling	1600	8.11036	8.033682	44
13	Control	Overstory	2550	8.072864	6.348897	48
13	Control	Sapling	5400	17.34966	6.395923	103
14	Tallow	Overstory	3250	6.809135	5.164863	44
14	Tallow	Sapling	3600	10.38244	6.059725	63
14	Control	Overstory	1200	39.15278	20.38193	147
14	Control	Sapling	2000	9.100466	7.61152	50
15	Tallow	Overstory	1100	15.57517	13.42687	69
15	Tallow	Sapling	2200	12.69815	8.572618	67
15	Control	Overstory	150	54.87085	68.2464	128
15	Control	Sapling	200	0.685069	6.603992	4
16	Tallow	Overstory	2150	1.955638	3.403138	14

Table 1.
Continued

Plot	Plot Type	Subplot	TPH Trees ha^{-1}	BA m^2ha^{-1}	QMD cm	RD %
16	Tallow	Sapling	2400	3.8074	4.494312	26
16	Control	Overstory	450	12.24864	18.61625	47
16	Control	Sapling	1600	40.2346	17.89346	159
17	Tallow	Overstory	2200	23.60573	11.68832	111
17	Tallow	Sapling	1600	4.96168	6.283606	29
17	Control	Overstory	1000	49.9649	25.22244	173
17	Control	Sapling	600	1.428915	5.506582	9
18	Tallow	Overstory	2150	18.19333	10.37986	89
18	Tallow	Sapling	2000	9.067024	7.597521	50
18	Control	Overstory	750	22.92243	19.72669	87
18	Control	Sapling	800	12.23699	13.95554	53
19	Tallow	Overstory	1300	13.09433	11.32464	62
19	Tallow	Sapling	1800	11.5428	9.03595	60
19	Control	Overstory	1000	50.66593	25.39876	175
19	Control	Sapling	600	6.717928	11.93979	31
20	Tallow	Overstory	3000	6.82763	5.383056	43
20	Tallow	Sapling	5000	18.65596	6.892524	108
20	Control	Overstory	2000	26.94746	13.0978	121
20	Control	Sapling	1000	3.197324	6.3804	19
21	Tallow	Overstory	2300	5.244929	5.388408	33
21	Tallow	Sapling	3400	7.685739	5.364854	49
21	Control	Overstory	950	42.69289	23.92052	151
21	Control	Sapling	200	0.973892	7.873991	5
22	Tallow	Overstory	1050	3.85275	6.835107	22
22	Tallow	Sapling	1600	3.873272	5.551797	24
22	Control	Overstory	3000	16.8617	8.459499	90
22	Control	Sapling	1600	11.66238	9.63359	59
23	Tallow	Overstory	2050	10.66923	8.140371	57
23	Tallow	Sapling	3600	12.71937	6.707121	74
23	Control	Overstory	850	10.66163	12.63738	48
23	Control	Sapling	1000	2.919648	6.097051	17

Table 1.
Continued

Plot	Plot Type	Subplot	TPH Trees ha^{-1}	BA m^2ha^{-1}	QMD cm	RD %
24	Tallow	Overstory	2450	9.797442	7.135557	56
24	Tallow	Sapling	6200	7.042221	3.802888	51
24	Control	Overstory	2200	36.57719	14.54951	157
24	Control	Sapling	6200	4.358698	2.991831	35
25	Tallow	Overstory	900	16.36184	15.21421	69
25	Tallow	Sapling	600	12.42143	16.23546	51
26	Tallow	Overstory	450	0.350388	3.14864	2
26	Tallow	Sapling	400	0.0831	1.626392	0.8
26	Control	Overstory	600	21.72306	21.47036	80
26	Control	Sapling	400	4.647521	12.16285	21
27	Tallow	Overstory	800	21.36101	18.43828	83
27	Tallow	Sapling	800	8.973789	11.95081	41
27	Control	Overstory	600	30.10121	25.27382	104
27	Control	Sapling	600	14.00641	17.2402	56
28	Tallow	Overstory	500	5.635094	11.97899	26
28	Tallow	Sapling	3000	15.27115	8.050629	83
28	Control	Overstory	750	24.13929	20.24353	91
28	Control	Sapling	800	2.86087	6.747745	16
29	Tallow	Overstory	650	5.510697	10.38966	27
29	Tallow	Sapling	800	1.308319	4.563167	8
29	Control	Overstory	650	34.7538	26.0915	119
29	Control	Sapling	1000	10.99758	11.83322	51
30	Tallow	Overstory	700	11.70697	14.59244	50
30	Tallow	Sapling	1400	9.861541	9.470282	50
30	Control	Overstory	750	16.4837	16.72828	67
30	Control	Sapling	2000	18.81912	10.94559	90
31	Tallow	Overstory	800	2.479066	6.281359	14
31	Tallow	Sapling	1000	10.29528	11.44916	48
31	Control	Overstory	600	37.84091	28.33738	125
31	Control	Sapling	600	0.12769	1.646106	1
32	Tallow	Overstory	1500	44.43141	19.42021	171

Table 1.
Continued

Plot	Plot Type	Subplot	TPH Trees ha^{-1}	BA m^2ha^{-1}	QMD cm	RD %
32	Tallow	Sapling	800	11.84581	13.73068	52
32	Control	Overstory	750	22.21456	19.41971	85
32	Control	Sapling	1200	7.276319	8.786579	38

Vita

Nicklaus R. Langlois graduated from Cedar Park High School in June of 2017 and enrolled in Stephen F. Austin State University starting the same summer. He played on the football team for four seasons while earning his Batchelor of Science in Environmental Science degree with a concentration in planning and management and a minor in Forestry Wildlife Management. After graduation in May of 2021, he was accepted into the Masters of Science in Environmental Science program at Stephen F. Austin State University concentrating his studies in wetlands and wetland management. Throughout graduate school he served as a teaching assistant for intro to environmental science, wetland delineation and function, and dendrology courses. During two summers in graduate school, he worked as an environmental intern for Mitigation Resources of North America. Because of his hard work as an intern, he was awarded a full-time position as an Ecologist for Mitigation Resources of North America in June of 2023. He received the degree of Master of Science in Environmental Science in August of 2024.

Style manual designation: MLA

This thesis was typed by Nicklaus R. Langlois