

Submitted: January 7, 2015

Revised and Resubmitted: February 3, 2015

Accepted: February 11, 2015

Available Online: February 27, 2015

<http://dx.doi.org/10.1016/j.foreco.2015.02.013>

The impact of Chinese tallow (*Triadica sebifera*) on stand dynamics in bottomland hardwood forests

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ABSTRACT

Chinese tallow (*Triadica sebifera* (L.) Small) is an invasive tree species that competes with native species in bottomland hardwood forests in the western Gulf Coastal Plain of the southern United States. While much research has been done on Chinese tallow in coastal prairie ecosystems focusing on its establishment ecology, there is little known about its impacts on stand dynamics in forested ecosystems, where it is a growing problem. A paired-plot design was employed in 23 different stands along the Neches River, near Diboll, Texas, to compare sites with contrasting abundance in Chinese tallow. The objectives were to 1) determine the impacts of Chinese tallow on stand dynamics, 2) examine its impact on the light environment, and 3) determine if edaphic factors were correlated with stand structure on sites with varying stocking of Chinese tallow. Stand structural metrics were measured in 2012 and 2013, and the below-canopy light environment was estimated using hemispherical canopy analysis. Soil samples were collected for analysis in 2013. For all species, stand density was greater in plots with abundant Chinese tallow, while basal area, quadratic mean diameter, and relative density were lower ($p < 0.10$). Chinese tallow presence was negatively correlated with overstory diversity, density of native species, oak density. We observed a greater maximum stand density index for bottomland hardwoods than has previously been reported (1200 trees ha^{-1} at 25 cm QMD). Stand structure was most correlated to soil metal availability, yet relationships were weak ($R^2 < 0.30$), potentially indicating minimal differences in edaphic conditions across our stands. Management to prevent Chinese tallow from replacing native species in bottomland hardwood forests includes proper density management of stands in areas where Chinese tallow is likely to grow and early detection of gap formation to allow treatment of establishing Chinese tallow before it outcompetes regenerating native species.

Keywords: invasive species, stocking, stand density management diagram, light environment, wetland

Highlights:

- Tallow abundance is correlated with lower overall stand stocking in bottomlands
- Tallow abundance is correlated with lower stocking of native species in bottomlands
- Tallow-dense areas have less overstory diversity but similar understory diversity
- Tallow presence did not alter below-canopy light quantity or gap fraction
- Soil factors did not correlate with tallow abundance in our study area

1. Introduction

The invasion of forested ecosystems by introduced exotic species has become a topic of concern for the ecological function of forests worldwide (Reichard and Hamilton, 1997; Richardson, 1998). One such species, Chinese tallow (*Triadica sebifera* (L.) Small), has increased in abundance by several orders of magnitude over the last century in bottomland hardwood forests in east Texas, altering ecosystems including coastal prairies and forested wetlands (Wall and Darwin, 1999; Hall and Harcombe, 2001). Evidence suggests Chinese tallow that is invasive in the United States displays differences in population genetics when compared to current-day populations found in China which may have allowed it to become more invasive since its widespread introduction in the Gulf Region in the early 1900's (Siemann and Rogers, 2003c). Compared to currently non-invasive Chinese ecotypes, the invasive ecotype found in the United States allocates more nitrogen to growth and reproduction and less to compounds that confer protection from herbivory. Additionally, its ability to sprout at rapid rates, produce a prolific crop of seeds at a young age, and tolerate a wide range of site conditions enable it to establish and thrive in the ecosystems of the southern United States (Bruce *et al.*, 1997; Barrilleaux and Grace, 2000). Known to convert grasslands and forests into Chinese tallow monocultures (Bruce *et al.*, 1995), it is extremely difficult to eliminate Chinese tallow once it becomes established (Miller *et al.*, 2010).

Bottomland oak flats, or low-elevation seasonally flooded backwater areas, are particularly prone to Chinese tallow invasion, as it often establishes at high densities (Wall and Darwin, 1999; Fan *et al.*, 2012). Native oak species are adversely affected by interspecific competition, particularly low light-levels created by a dense mid-story (Lorimer, 1993; Dey, 2002). By contrast, Chinese tallow may be better suited to survive and grow despite competing vegetation. Although Chinese tallow is able to grow under varying light environments, it performs better in full sunlight (Jones and McLeod, 1990; Barrilleaux and Grace, 2000; Battaglia *et al.*, 2009). Chinese tallow seedlings benefit from open canopy

habitats, allowing them to grow at faster rates on a range of sites (Pattison and Mack, 2009). However, its response to light may be moderated by other site characteristics beyond light availability. The combination of an open canopy and greater moisture availability are positively correlated with Chinese tallow growth (Lin *et al.*, 2004).

Although the species is able to survive in dry areas, mesic environments are most favorable to Chinese tallow for establishment (Jones and Sharitz, 1990). Chinese tallow may increase the amount hypertrophied lenticels and obtain thicker feeder roots as a response to establishing in areas with flood regimes (Jones and Sharitz, 1990; Conner, 1994; Conner *et al.*, 1997; Conner *et al.*, 2001). Additionally, flooding duration does not seem to affect Chinese tallow survival on floodplains in Texas (Siemann and Rogers, 2003b). Thus, it is able to compete with native bottomland hardwood species. The probability of invasion of forests by Chinese tallow increased with a decreasing distance to a body of water (Gan *et al.*, 2009), which is attributable to water being one of the dominant means by which Chinese tallow seed dispersal occurs.

Chinese tallow growth rates are also affected by soil nutrient availability; increased nitrogen, potassium, and phosphorus result in better Chinese tallow growth (Siemann and Rogers, 2003a). Increased nutrition allows for improved height, diameter, aboveground biomass, shoot mass and root mass (Rogers and Siemann, 2002). Through its rapid leaf decomposition rate, Chinese tallow also increases nutrient cycling rates in its immediate environment, self-facilitating its own growth (Cameron and Spencer, 1989). Interactions between light availability and nitrogen addition have been observed for Chinese tallow growth rates, also resulting in morphological variability (Siemann and Rogers, 2003a).

However, whether many of the previous studies on Chinese tallow are applicable at the stand scale in bottomland hardwoods remains in question, as most were conducted in grassland ecosystems, as short-term field experiments, or as greenhouse studies (Cameron *et al.*, 2000; Nijjer *et al.*, 2002; Rogers and

Siemann, 2003; Siemann and Rogers, 2003a; Donahue *et al.*, 2006; Zou *et al.*, 2009). More research is needed on the impact of already established Chinese tallow on mature and aggrading bottomland hardwood forests following stand initiation. The data presented in this study directly address this research gap. To evaluate and compare differences between areas of high and low Chinese tallow abundance, a paired-plot design was established within bottomland oak dominated forests located in the Pineywoods Mitigation Bank near Diboll, Texas. Differences in stand structure, stocking, composition, light environment, and edaphic factors were examined over two growing seasons, 2012 and 2013. We hypothesized that (i) areas with a greater Chinese tallow density would have a reduced density of native species and reduced overall stand stocking; (ii) due to Chinese tallow's rapid growth rates, areas with more Chinese tallow would have an altered below-canopy light environment; and (iii) soil hydrology within bottomland hardwood forests would not be correlated with Chinese tallow density, but fertile soils with greater levels of plant available nutrients would prove more conducive to its growth relative to native species.

2. Materials and methods

2.1 Study area

The Pineywoods Mitigation Bank (PMB) is a 7,721 ha forested wetland located along the Neches River in east Texas (Figure 1). Average annual rainfall within the region is 147 centimeters with an average high of 34 °C and low of 22 °C in the summers and a high of 16 °C and low of 3 °C in the winters. Primarily dominated by mixed hardwood forests, the PMB has abundant willow oak (*Quercus phellos*), laurel oak (*Q. laurifolia*), and water oak (*Q. nigra*) in mesic flats. Within the swamps, black tupelo (*Nyssa sylvatica*) and bald cypress (*Taxodium distichum*) are the dominant species. Other abundant species include green ash (*Fraxinus pennsylvanica*), American elm (*Ulmus americana*), and sweetgum (*Liquidambar*

styraciflua). Restoration of stand structure, composition, and ecological function is the current management objective for the PMB.

The spatial distribution of Chinese tallow is variable at the PMB; patch sizes range from a few square meters up to several hectares. Chinese tallow regeneration often occurs in irregularly shaped areas of blow-down within mature stands. Of the total of 7,721 hectares, 5,261 hectares of bottomland forest are in the process of being treated with herbicides to control Chinese tallow, although treated areas have been excluded from this study. Herbicide treatments have been performed since 2009, using basal spray and foliar applications of Garlon® (triclopyr), foliar application of Clearcast™ (imazamox), and stem injection of Arsenal® (imazapyr). All applications have had varying degrees of success based on the specific area in the bank where the Chinese tallow established. Approximately 1,000 hectares will be treated annually until 2016.

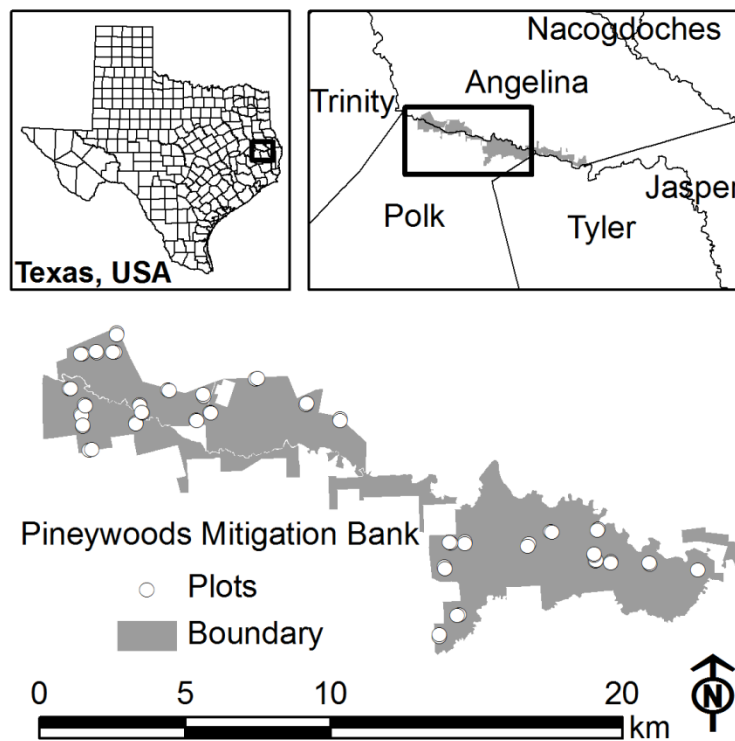


Figure 1. Map of the Pineywoods Mitigation Bank in Angelina, Polk, Tyler, and Jasper Counties, Texas, USA.

2.2. Experimental design

A paired-plot experimental design (e.g. Stape *et al.*, 2004) was installed throughout the study area (Figure 1). Plots were selected so one would have little to no tallow (native plot) while the other in the pair would have abundant Chinese tallow (tallow plot). Plots were replicated among stands (i.e. stands are the experimental unit), but not within each stand (i.e. only one pair of plots per stand). In this study, stands were considered to be contiguous areas with relatively homogeneous composition, stand structure, and soils. However, they were not completely homogeneous, as recent developments in the silvicultural literature are more accepting of within-stand heterogeneity (O'Hara and Nagel, 2013). Stands were previously defined by the Working Lands Investment Partners as management units during winter and spring of 2012. Paired-plots were separated from other pairs by a minimum distance of 305 meters. Identification of a Chinese tallow plot was based on Chinese tallow density. To be considered a Chinese tallow plot, a stand must have met at least one of the following criteria: 2 Chinese tallow trees per 200 m² plot > 25.4 cm dbh, 4 trees per 200 m² plot > 17.8 cm dbh, 8 trees per 200 m² plot > 10.2 cm dbh, 16 trees per 50 m² plot > 2.5 cm dbh, or 32 trees per 50 m² plot < 2.5 cm dbh. These criteria were determined based on tallow densities observed at the PMB on several pilot plots and were chosen so that tallow plots would be consistent with areas that would be later targeted for herbicide application to meet management objectives.

Once an area within a stand which met the above criteria was identified as the tallow plot, the native paired-plot was then located and installed in an area of the stand which did not meet the above criteria, and thus had little Chinese tallow present and would not later be targeted for herbicide application. A random distance between 32 and 96 meters at a random azimuth was used between tallow and native plots. New random distances and azimuths were repeated from each tallow plot until an acceptable native plot (i.e. one that was within the same stand but did not meet above tallow plot criteria) was located. The distance range between paired-plots was selected based on stand sizes at the study site. A

total of 32 paired-plots were installed. Plots were measured once each year for two years, in 2012 and 2013. Two plots were not measured in the first year, due to access issues, so these data are excluded from analysis. Between the first and second field seasons, herbicide treatments were conducted within some of the plots. Since we did not have a sufficient number of such plots at this time (only 7), and were only able to measure them in the first year following application, herbicide effects are not considered in this manuscript. As such, treated plots were removed, leaving 23 pairs of plots to be analyzed with two years of data collection.

2.3. Data collection

2.3.1. Stand structure

Within each tallow and native plot, three sets of nested sampling subplots were installed from plot center: one overstory, one sapling, and four seedling subplots. The circular overstory and sapling subplots were installed at the plot center. The overstory subplots were 200 m², while the sapling subplots were 50 m². The four seedling subplots were 1 m² in area, and located 3 m from plot center in each cardinal direction. Any tree species > 10.2 cm dbh was tallied as an overstory tree. Tree species between 2.5 and 10.2 cm dbh were tallied as saplings. Finally, seedling size was anything less than 2.5 cm dbh. Vines and shrubs were also sampled in seedling subplots. In each overstory and sapling subplot, trees (live and dead) > 10.2 cm dbh were measured during the 2012 and 2013 growing seasons. Density, basal area, quadratic mean diameter (QMD), stand density index (SDI) (Reineke, 1933), and relative density were calculated for the 23 plots in both the 2012 and 2013 growing seasons. Briefly, the SDI of a stand describes an identically stocked stand that has a QMD of 25 cm and a density (trees ha⁻¹) given by the value of SDI (see Reineke, 1933). This number is compared to a characteristic maximum SDI for that species and region, from which inferences about stocking can be made. The maximum SDI approximates the size-density combination at which carrying capacity is reached for that species. The value of SDI is that it allows stocking comparisons to be made across stands with a wide range of

densities and QMD's. A maximum SDI of 570 was used to calculate relative density (Schnur, 1937).

While Schnur's data are for upland hardwoods, this was the most comparable ecosystem with published data.

All woody species were identified for the overstory, sapling, and seedling subplots for both the 2012 and 2013 growing seasons. Species counts in the overstory and understory were used to test differences in diversity between tallow and control plots. Species richness, Shannon's diversity index (H), and Shannon's evenness (E) were calculated (Peet, 1974; Molles and Cahill, 1999). Briefly, H ranges between 0 and a typical maximum of 4.0, with greater values representing more diverse communities. E ranges between 0 and 1, where a value of 1 indicates that there is the same number of individuals of each species within a community, or that it is more even.

2.3.2. *Light environment*

Hemispherical canopy photography was used to estimate canopy cover, direct light and indirect light transmitted through the canopy, and leaf area index (LAI). Hemispherical canopy photography is an accurate technique for studying plant canopies (Evans and Coombe, 1959; Rich, 1990). Using a hemispherical (fisheye) lens underneath a canopy, a permanent record of canopy structure can be created, including such factors as the position, size, density and distribution of canopy gaps (Rich, 1990; Hale and Edwards, 2002). This equi-angular, 180 degree, projection displays a complete view of all sky directions, with the zenith at center and horizons at photograph edges. The photos were interpreted by classifying pixels as either sky or canopy, allowing for the estimation of potential solar radiation penetration through openings based on known sun-tracks throughout the year (Evans and Coombe, 1959). Through the photo and sun-tracks, it was possible to create indices of the amount of direct light (light propagated directly from the disk of the sun without reflecting) and diffuse light (light reflecting off clouds, canopy elements, etc.) that were transmitted through the canopy to the location of the

camera. These are termed direct site factor (DSF) and indirect site factor (ISF), respectively. LAI is single-sided leaf area per unit ground surface area. This was estimated from hemispherical photographs by using an inversion model based on Beer's Law using the observed gap fraction distribution throughout the photo (Rich, 1990; Rich *et al.*, 1999).

Hemispherical photographs were taken in the center of each plot, using a SIGMA SD15 digital camera with a 4.5 mm 1:2.8 fisheye lens (SIGMA Japan). The camera was mounted on a tripod set 1.3 m above the ground and leveled so the center of the lens was focused on the zenith and a marker to magnetic north was identifiable. Photographs were taken on each plot between July 1, 2012, and July 10, 2012. A second set of photographs was taken between July 1, 2013 and August 12, 2013. All photographs were taken pre-dawn, post-dusk, or on overcast days when cloud cover prevented direct sunlight from creating errors in interpretation by minimizing reflected light from leaves and boles and increasing contrast between sky and canopy. Photographs were analyzed using SideLook version 1.1 to determine thresholds for pixel classification (Nobis and Hunziker, 2005). SideLook is a software program used for analyzing vegetation structure from digital images. Each pixel in the image was classified as either sky or canopy. Once thresholds were established, Hemiview canopy analysis software (Delta-T Devices, England) was used to calculate below-canopy light indices and LAI for each photo.

2.3.3. Edaphic factors

To assess soil water table depth, a piece of un-rusted steel rebar was installed to a depth of 50 cm within 3 m of plot center in summer 2012. They were removed and checked periodically throughout the next year. At each removal, they were examined for the depth of the rust line, as portions covered by the water table were unable to be oxidized and rust (Bridgham *et al.*, 1991).

Three soil samples were collected from random locations within each plot using a shovel, at a depth between 0 cm to 10 cm (Schneider and Sharitz, 1986). Subsamples were mixed within a bucket to

homogenize soil samples. Samples were then placed in sealable bags and transported to the Soil, Plant, and Water Analysis Laboratory at Stephen F. Austin State University where soil analyses were performed. The bulked samples were fan dried for two days and then ground to prepare them. A Mehlich 3 extraction procedure was used to test for plant-available macronutrient availability with an IRIS Intrepid II XSP inductively coupled plasma (ICP) analyzing unit (Thermo Scientific, USA) (Mehlich, 1984). For total soil metals, the EPA method 3050B was employed (Edgell, 1989). This procedure involved acid digestion to obtain an analyte which was passed through a Spectronic GENESYS 10 Vis Spectrophotometer (Thermo Scientific, USA) to determine elemental concentrations in a sample. Additionally, soil pH and electrical conductivity tests were conducted by mixing soil samples with deionized water and then measuring for acidity and sodium concentration. Carbon and nitrogen mass percentage were analyzed with a Vario MACRO Macro Elemental Analyzer (Elementar, Germany). Soils were further analyzed to determine texture using the hydrometer test (Gee and Bauder, 1979). A 4% sodium hexa-metaphosphate solution was mixed with the soil sample and water in a graduated cylinder. As time progressed, the sand, silt, and clay separated, allowing for the determination of textural distribution within the sample through the use of a hydrometer.

2.4. Data analysis

SAS 9.3 software was used to perform statistical tests (SAS Institute, Cary NC). A two way analysis of variance (PROC GLM) was done to determine whether there was a significant difference in stand structural metrics (density, basal area, QMD, relative density) between plot types (native versus tallow), year of measurement, or any interactions ($p < 0.10$). The same test was also performed for light environment data. To meet test assumptions, transformations were made when necessary to meet the Shapiro-Wilk normality test. Both Levine's test and residual plots were examined to confirm constant variance.

Regressions (PROC REG) comparing native plots (x-axis) to Chinese tallow plots (y-axis) were performed to examine differences in stand structure and edaphic factors between areas with and without Chinese tallow. By examining the distribution of data relative to a line with a slope of 1.0 and intercept at the origin, it was possible to quickly determine the impacts of increased Chinese tallow density on stand structure (see Figure 2 for an example). By examining the statistics from these regressions, it was also possible to determine the extent to which native plot and Chinese tallow stand structure were correlated. As with the ANOVA, transformations were performed as necessary on all data to meet normality assumptions, while residual plots were examined to test for heteroscedasticity.

A Principal Component Analysis (PCA) was performed on the edaphic site variables utilizing PC-ORD program version 6.08 (Kooch *et al.*, 2008). PCA, an ordination technique introduced by Pearson, is a tool utilized to examine the structure of data. PCA allowed us to select variables that accounted for the greatest variability in our dataset so that these could be used in subsequent analyses of the relationship between stand structure and edaphic factors without overfitting models.

3. Results

3.1. Stand structure

Chinese tallow and native plots differed in stand density, basal area, QMD, and relative density across all species ($p < 0.10$, Tables 1, 2). Density was higher in Chinese tallow plots for all species, while basal area, QMD, and relative density were lower (Figure 2). This is demonstrated graphically where data are clustered on one side of the 1:1 line. Density and relative density also varied between the 2012 and 2013 growing seasons ($p < 0.10$, Tables 1, 2). No plot-type-by-year interactions were observed. Density lessened between the two growing seasons from a mean of 2,335.9 trees per hectare in 2012 to a mean of 1,445.7 per hectare in 2013, while relative density dropped from 108.8% to 87.7%. Table 2 also

demonstrates that the plot selection protocols were effective, as native plots had a minimal presence of tallow in both years, particularly when compared to tallow plots.

Table 1. Two-way ANOVA results for stand structural metrics between native and Chinese tallow plots for 2012 and 2013. Significant values are bolded. Relative density was calculated assuming a maximum SDI of 570 based on Schnur (1937).

Effect	Density	Basal Area	QMD	Rel. Density
Plot Type	0.0296	0.0001	0.0001	0.0006
Year	0.0055	0.1029	0.4591	0.0263
Plot Type * Year	0.1834	0.4953	0.5405	0.4675

Table 2. Mean and standard errors (in parentheses) for stand structural variables for 23 pairs of plots measured over two growing seasons along the Neches River near Diboll, Texas. Relative density was calculated assuming a maximum SDI of 570 based on Schnur (1937).

			Density (trees ha ⁻¹)	Basal Area (m ² ha ⁻¹)	QMD (cm)	Rel. Density (%)
Native Plots	2012	Native Species	1767.4 (324.1)	29.0 (2.2)	19.2 (2.4)	---
		<i>Triadica sebifera</i>	39.1 (22.2)	0.2 (0.1)	1.5 (0.9)	---
		All Species	1815.2 (331.4)	29.4 (2.3)	19.0 (2.4)	122.5 (9.5)
	2013	Native Species	1339.1 (236.9)	26.4 (2.7)	18.8 (2.4)	---
		<i>Triadica sebifera</i>	37.0 (34.8)	0.1 (0.1)	1.1 (0.9)	---
		All Species	1376.1 (229.9)	26.5 (2.7)	18.9 (2.3)	108.5 (10.9)
Tallow Plots	2012	Native Species	847.8 (207.8)	10.0 (1.9)	13.9 (1.4)	---
		<i>Triadica sebifera</i>	2130.4 (369.8)	8.9 (1.2)	9.3 (1.4)	---
		All Species	2856.5 (355.4)	18.7 (2.2)	9.7 (1.0)	95.2 (9.8)
	2013	Native Species	539.1 (143.6)	10.4 (3.0)	14.5 (1.9)	---
		<i>Triadica sebifera</i>	976.1 (210.9)	4.1 (0.8)	5.7 (0.9)	---
		All Species	1515.2 (193.1)	14.5 (2.9)	11.6 (1.3)	67.0 (10.3)
All Plots	2012	Native Species	1307.6 (202.3)	19.5 (2.0)	16.5 (1.4)	---
		<i>Triadica sebifera</i>	1084.8 (240.5)	4.5 (0.9)	5.4 (1.0)	---
		All Species	2335.9 (252.5)	24.0 (1.8)	14.3 (1.4)	108.8 (7.0)
	2013	Native Species	919.1 (147.5)	18.0 (2.3)	16.6 (1.6)	---
		<i>Triadica sebifera</i>	495.7 (124.5)	2.1 (0.5)	3.4 (0.7)	---
		All Species	1445.7 (148.8)	20.5 (2.1)	15.3 (1.4)	87.7 (8.0)

Regressions revealed that stand structure was not correlated between tallow and native plots for any of the four metrics assessed ($p > 0.10$; Figure 2). Oaks were more abundant in native plots while in Chinese tallow plots oaks were less dominant by all metrics (Figure 3). Statistics for the oak species were unable

to be tested parametrically or non-parametrically due to the strongly skewed nature of the data because a large number of tallow plots had no oak component.

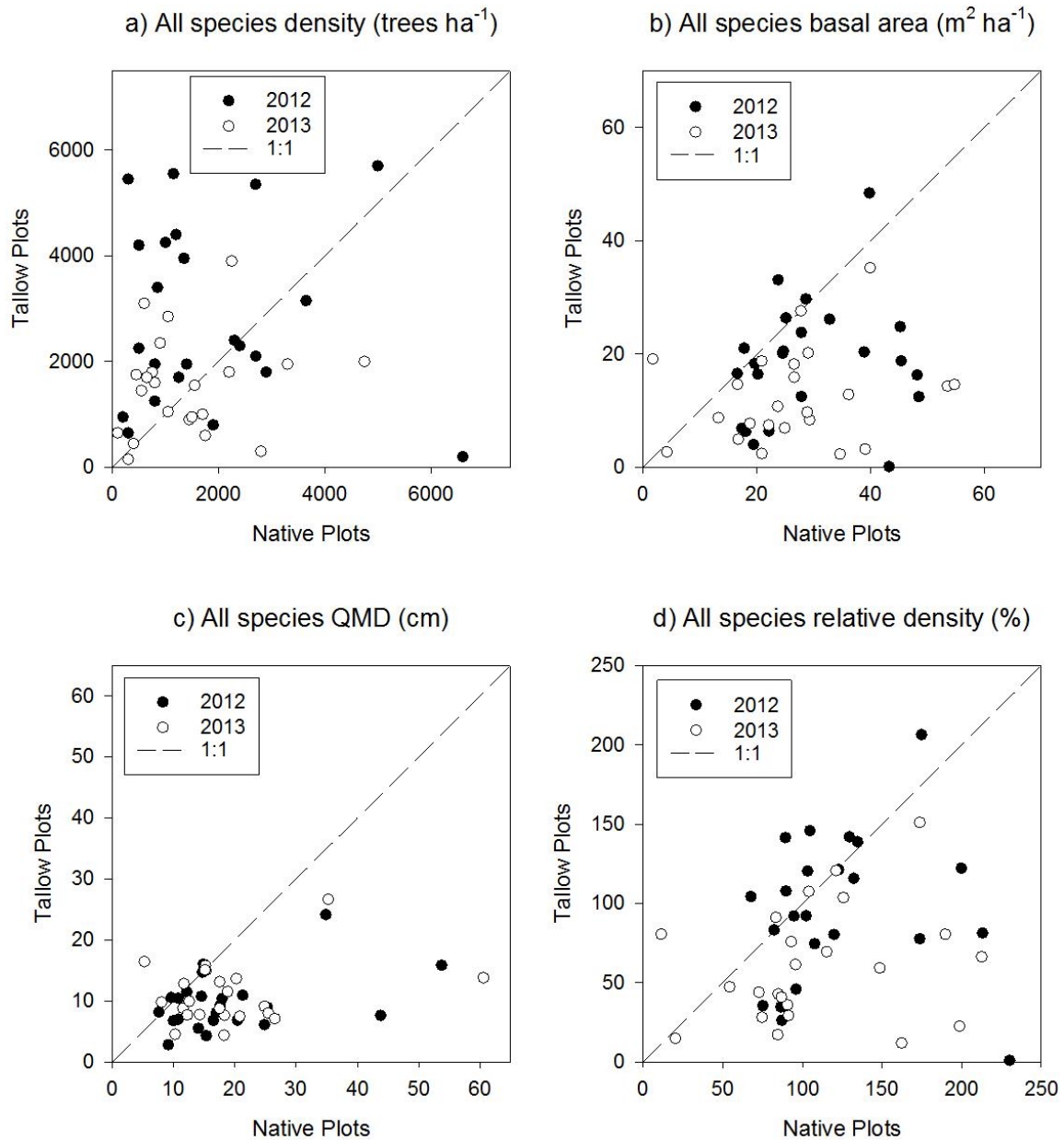


Figure 2. Comparisons for all species between native plots and Chinese tallow plots in the paired-plot design located at the Pineywoods Mitigation Bank near Diboll, Texas. A dotted 1:1 line (slope = 1.0) is included on each graph for reference. Points below the 1:1 line have a greater value of the variable in native plots, while points above the 1:1 line have a greater value in the Chinese tallow plots.

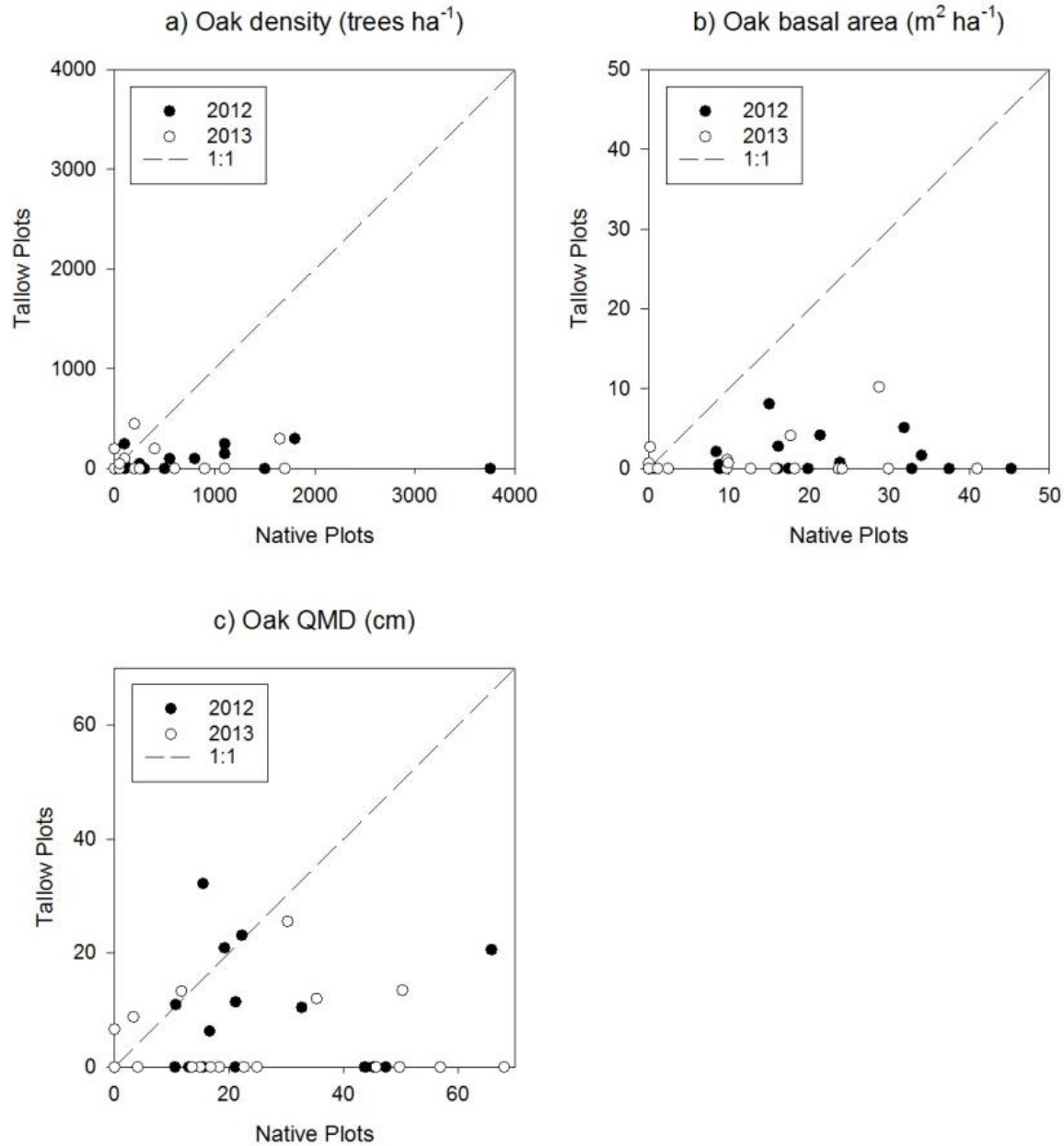


Figure 3. Comparisons for oak species between native plots and Chinese tallow plots in the paired-plot design located at the Pineywoods Mitigation Bank near Diboll, Texas. A dotted 1:1 line (slope = 1.0) is included on each graph for reference. Points below the 1:1 line have a greater value of the variable in native plots, while points above the 1:1 line have a greater value in the Chinese tallow plots.

Stocking of all species in almost all plots was high in both years (Table 2, Figure 4), with a visually estimated maximum SDI line observed in these stands of approximately 1200 trees ha⁻¹ at a QMD of 25.0 cm (485 in trees acre⁻¹ at a QMD of 10.0 inches). Additionally, it appears the exponent (shown in Figure 4 legend) diverged from the 1.605 calculated by Reineke (1933), and was approximated at 1.750.

This is not unexpected, as this exponent does vary by species (Pretzsch and Biber, 2005; Pretzsch, 2006). For comparison, maximum SDI for mixed hardwood species from the central hardwoods region of the United States is frequently reported at between 530 – 620, substantially lower than what was observed in our stands (Gingrich, 1967; Williams, 2003; Johnson *et al.*, 2009). The trend of Chinese tallow plots having smaller QMD but a greater density is also readily apparent in the stand density management diagram (Figure 4), as the dotted lines for Chinese tallow plots are clustered to the lower right, while the solid lines for the native plots are clustered closer to the maximum SDI line.

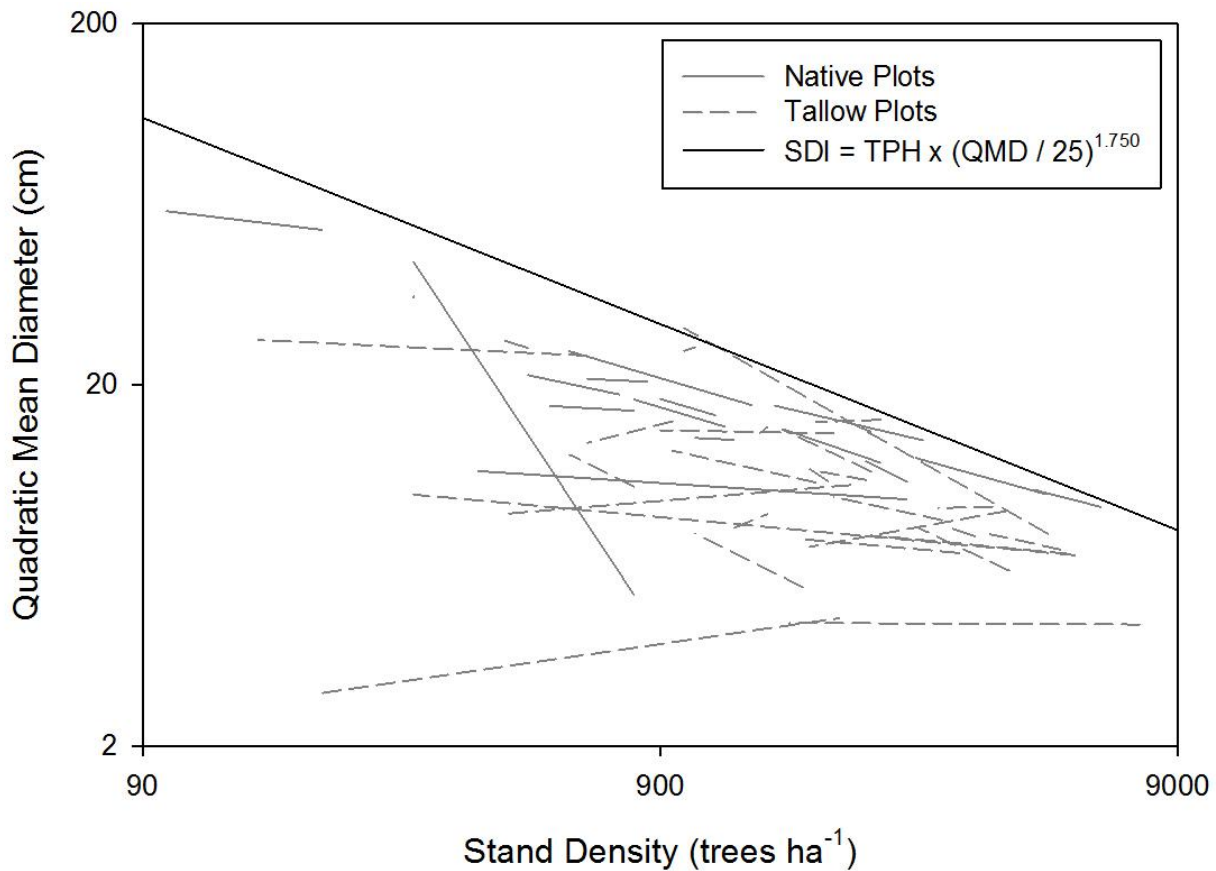


Figure 4. A simplified stand density management diagram for all measured Chinese tallow and native plots. Note that the scale on both axes is logarithmic. The solid black line represents an estimated maximum SDI of 1200 trees ha⁻¹ at a QMD of 25 cm, which is equivalent to 485 trees acre⁻¹ at a QMD of 10 inches.

3.2. Light environment and diversity

No differences in the light environment existed between native and tallow plots, sampling year, or interactions for percent visible sky, direct, or indirect site factors (Tables 3 and 4). However, Chinese tallow plots did have a greater mean leaf area index, although the effect was slight in terms of magnitude (2.6 versus 2.3). There were no interaction effects.

Table 3. Two-way ANOVA results for light environment between pair-plots for 2012 and 2013. The only significant value ($p < 0.10$) is bolded.

	Visible Sky	Indirect Site Factor	Direct Site Factor	Leaf Area Index
Plot Type	0.85	0.36	0.17	0.08
Year	0.26	0.18	0.33	0.68
Plot Type * Year	0.64	0.64	0.94	0.95

Table 4. Mean and standard errors (in parentheses) for light environment variables estimated using hemispherical canopy photography on 23 pairs of plots for two growing seasons along the Neches River near Diboll, Texas.

		Visible Sky (%)	Indirect Site Factor (%)	Direct Site Factor (%)	Leaf Area Index ($m^2 m^{-2}$)
Native Plots	2012-13	12.3 (0.9)	15.6 (1.3)	15.8 (1.3)	2.3 (0.1)
	2012	13.8 (1.5)	18.1 (2.0)	17.9 (2.1)	2.2 (0.2)
	2013	10.7 (1.1)	13.1 (1.4)	13.6 (1.3)	2.4 (0.1)
Tallow Plots	2012-13	12.2 (1.1)	16.9 (1.6)	18.1 (1.8)	2.6 (0.1)
	2012	13.6 (1.9)	19.1 (2.7)	20.9 (3.0)	2.6 (0.2)
	2013	10.9 (1.1)	14.7 (1.5)	15.2 (1.9)	2.7 (0.2)
All Plots	2012-13	12.2 (0.7)	16.3 (1.0)	16.9 (1.1)	2.5 (0.1)
	2012	13.7 (1.2)	18.6 (1.7)	19.5 (1.8)	2.4 (0.1)
	2013	10.8 (0.8)	13.9 (1.0)	14.4 (1.2)	2.5 (0.1)

The overstory had greater species richness and diversity (H) in native plots as compared to tallow plots (Tables 5, 6). While evenness did not differ between plot types, it did increase slightly (0.62 to 0.70) between 2012 and 2013. We observed no differences in understory diversity across plot types or between growing seasons (Tables 5, 7). However, understory density of native species was 162%

greater in native plots than tallow plots. The growing seasons also differed, with a 45% reduction in native density and a 39% reduction in density of all species from 2012 to 2013.

Table 5. Two-way ANOVA results for overstory diversity and understory diversity and density data between pair-plots for 2012 and 2013. Only significant values ($p < 0.10$) are bolded. Because of the absence of tallow in many native plots, it was not possible to transform tallow density to normal. P-values for that variable are derived from separate Wilcoxon Rank Sum Tests for each main effect. The interaction was not tested.

	Overstory			Understory					
	Richness	H	E	Richness	H	E	Native Dens	Tallow Dens	All spp Dens
Plot Type	0.07	0.01	0.14	0.93	0.93	0.34	0.08	<0.01	0.58
Year	0.39	0.82	0.09	0.25	0.71	0.77	0.01	0.53	0.04
Plot Type * Year	0.31	0.90	0.23	0.51	0.48	0.90	0.31	N/A	0.50

Table 6. Mean and standard errors (in parentheses) for overstory diversity for 23 pairs of plots over two growing seasons along the Neches River near Diboll, Texas. H is Shannon's diversity index, while E is evenness. Greater values correlate with greater species diversity more evenly distributed amongst all species.

		Richness (# species)	H (unitless)	E (unitless)
Native Plots	2012-13	4.30 (0.23)	0.96 (0.07)	0.70 (0.03)
	2012	4.27 (0.34)	0.94 (0.10)	0.69 (0.05)
	2013	4.32 (0.32)	0.98 (0.09)	0.71 (0.05)
Tallow Plots	2012-13	3.69 (0.26)	0.72 (0.07)	0.62 (0.04)
	2012	4.00 (0.38)	0.71 (0.09)	0.55 (0.05)
	2013	3.36 (0.35)	0.72 (0.10)	0.69 (0.06)
All Plots	2012-13	3.99 (0.18)	0.84 (0.05)	0.66 (0.03)
	2012	4.13 (0.25)	0.82 (0.07)	0.62 (0.04)
	2013	3.84 (0.25)	0.85 (0.07)	0.70 (0.04)

Table 7. Mean and standard errors (in parentheses) for understory diversity and density for 23 pairs of plots over two growing seasons along the Neches River near Diboll, Texas. H is Shannon's diversity index, while E is evenness. Greater values correlate with greater species diversity more evenly distributed amongst all species.

		Richness (# species)	H (unitless)	E (unitless)	Native (stems m ⁻²)	Tallow (stems m ⁻²)	All Species (stems m ⁻²)
Native Plots	2012-13	7.83 (0.49)	1.55 (0.08)	0.82 (0.02)	9.93 (1.60)	0.55 (0.25)	10.48 (1.64)
	2012	8.00 (0.65)	1.56 (0.10)	0.82 (0.03)	12.86 (2.96)	0.62 (0.46)	13.48 (3.02)
	2013	7.65 (0.74)	1.53 (0.13)	0.83 (0.03)	7.01 (0.99)	0.48 (0.19)	7.49 (1.02)
Tallow Plots	2012-13	7.89 (0.49)	1.51 (0.09)	0.77 (0.03)	6.14 (0.60)	4.23 (1.11)	10.37 (1.18)
	2012	8.52 (0.66)	1.57 (0.13)	0.77 (0.04)	7.84 (0.96)	4.54 (1.49)	12.38 (1.58)
	2013	7.26 (0.71)	1.44 (0.13)	0.77 (0.04)	4.43 (0.56)	3.92 (1.68)	8.36 (1.70)
All Plots	2012-13	7.86 (0.34)	1.53 (0.06)	0.79 (0.02)	8.04 (0.88)	2.39 (0.60)	10.43 (1.01)
	2012	8.26 (0.46)	1.57 (0.08)	0.79 (0.02)	10.35 (1.58)	2.58 (0.83)	12.93 (1.69)
	2013	7.46 (0.51)	1.48 (0.09)	0.80 (0.03)	5.72 (0.59)	2.20 (0.87)	7.92 (0.98)

3.3. Edaphic factors

Most edaphic factors showed no correlation to stand structural variables across all plots or within Chinese tallow or native plots only (Table 8). None of the correlations observed were large ($R^2 < 0.30$), although some did explain a small amount of the variability observed in stand structure. For instance, in the Chinese tallow plots pH was negatively correlated to all species QMD ($R^2 = 0.12$), all species basal area ($R^2 = 0.15$), native species QMD ($R^2 = 0.27$), and native species basal area ($R^2 = 0.13$). However, there were no correlations in the native plots. Clay content and field capacity were weakly correlated to leaf area index in both Chinese tallow and control plots, while other correlations were less consistent between plot types.

Table 8. P-values for simple linear regressions between edaphic variables and stand structure or light variables for both native and tallow plots measured along the Neches River in east Texas. Significant values ($p < 0.10$) are shown in bold font.

Variable	Chinese Tallow			Native Species			All Species			Light Environment				
	Dens	BA	QMD	Dens	BA	QMD	Dens	BA	QMD	VisSky	ISF	DSF	LAI	
Native Plots	Clay	0.920	0.661	0.575	0.816	0.849	0.360	0.822	0.859	0.348	0.307	0.355	0.829	0.044
	Sand	0.355	0.464	0.765	0.165	0.354	0.853	0.198	0.362	0.914	0.458	0.427	0.883	0.105
	Silt	0.144	0.154	0.393	0.021	0.057	0.699	0.033	0.063	0.604	0.722	0.618	0.951	0.352
	Field Capacity	0.664	0.856	0.939	0.596	0.737	0.590	0.631	0.739	0.609	0.368	0.389	0.815	0.070
	pH	0.942	0.921	0.835	0.216	0.767	0.447	0.206	0.767	0.416	0.066	0.069	0.585	0.825
	Elec. Cond.	0.500	0.321	0.376	0.455	0.027	0.518	0.234	0.029	0.519	0.252	0.165	0.325	0.203
	C	0.289	0.606	0.090	0.507	0.765	0.320	0.600	0.750	0.403	0.676	0.774	0.710	0.653
	N	0.634	0.206	0.018	0.755	0.888	0.333	0.803	0.920	0.365	0.619	0.646	0.682	0.763
	P	0.670	0.929	0.908	0.206	0.121	0.597	0.253	0.084	0.806	0.953	0.898	0.878	0.338
	K	0.693	0.812	0.898	0.532	0.273	0.562	0.640	0.224	0.621	0.948	0.872	0.855	0.117
	Ca	0.995	0.741	0.614	0.934	0.961	0.679	0.765	0.861	0.725	0.599	0.536	0.494	0.129
	Mg	0.850	0.670	0.594	0.760	0.988	0.702	0.683	0.976	0.604	0.467	0.449	0.347	0.020
	S	0.522	0.488	0.503	0.691	0.873	0.635	0.391	0.681	0.731	0.922	0.746	0.971	0.223
	Zn	0.574	0.975	0.670	0.739	0.900	0.590	0.982	0.900	0.632	0.072	0.143	0.108	0.065
	Tallow Plots	Clay	0.312	0.219	0.506	0.159	0.219	0.840	0.937	0.281	0.176	0.498	0.938	0.859
Sand		0.922	0.714	0.229	0.616	0.714	0.109	0.629	0.517	0.414	0.660	0.425	0.177	0.676
Silt		0.471	0.240	0.322	0.805	0.240	0.080	0.540	0.154	0.070	0.353	0.326	0.141	0.360
Field Capacity		0.547	0.346	0.338	0.763	0.816	0.440	0.948	0.973	0.932	0.546	0.964	0.713	0.038
pH		0.316	0.742	0.208	0.415	0.089	0.010	0.621	0.062	0.092	0.050	0.137	0.067	0.159
Elec. Cond.		0.899	0.972	0.564	0.483	0.053	0.351	0.471	0.471	0.771	0.058	0.050	0.420	0.368
C		0.852	0.666	0.998	0.175	0.636	0.433	0.171	0.535	0.520	0.461	0.350	0.089	0.942
N		0.709	0.754	0.856	0.134	0.714	0.465	0.821	0.770	0.888	0.510	0.445	0.092	0.707
P		0.398	0.522	0.697	0.797	0.252	0.992	0.504	0.308	0.751	0.493	0.512	0.052	0.643
K		0.391	0.429	0.465	0.132	0.088	0.586	0.614	0.112	0.202	0.291	0.447	0.490	0.038
Ca		0.583	0.344	0.210	0.570	0.975	0.306	0.398	0.883	0.831	0.187	0.226	0.499	0.140
Mg		0.591	0.442	0.303	0.263	0.415	0.565	0.248	0.289	0.666	0.402	0.527	0.827	0.125
S		0.869	0.468	0.481	0.357	0.386	0.114	0.887	0.789	0.906	0.134	0.076	0.013	0.888
Zn		0.621	0.692	0.015	0.020	0.145	0.586	0.209	0.278	0.472	0.798	0.679	0.754	0.338

Table 9. Descriptive data including mean, standard error, and range values for edaphic factors observed on 23 pairs of plots along the Neches River in east Texas.

Values	Native Plots		Tallow Plots		All Plots
	Mean (SE)	(Min, Max)	Mean (SE)	(Min, Max)	Mean (SE)
Clay (%)	22.6 (1.7)	[9, 39]	23.6 (1.3)	[11, 37]	23.1 (1.1)
Sand (%)	49.8 (3.5)	[16, 77]	46.3 (2.5)	[21, 67]	48.1 (2.2)
Silt (%)	27.6 (2.4)	[12, 59]	30.1 (2.2)	[14, 56]	28.8 (1.6)
Field Capacity	0.26 (0.01)	[0.17, 0.36]	0.27 (0.01)	[0.21, 0.35]	0.26 (0.01)
pH	4.9 (0.1)	[4.2, 5.4]	4.9 (0.0)	[4.4, 5.3]	4.9 (0.0)
E.C. (dS m ⁻¹)	342.2 (195.5)	[35.7, 4620.0]	671.3 (497.0)	[49.6, 11,590.0]	506.7 (265.2)
C (% by mass)	5.6 (0.7)	[1.1, 15.9]	6.0 (0.5)	[3.1, 11.9]	5.8 (0.4)
N (% by mass)	1.4 (0.1)	[0.6, 2.7]	1.5 (0.1)	[0.8, 3.0]	1.5 (0.1)
P (ppm)	5.6 (0.7)	[1.1, 15.9]	6.0 (0.5)	[3.1, 11.9]	5.8 (0.4)
K (ppm)	84.0 (8.5)	[30.8, 162.6]	83.9 (8.0)	[32.4, 177.4]	83.9 (5.8)
Ca (ppm)	658.3 (81.3)	[196.9, 1509.0]	754.7 (63.4)	[314.5, 1348.0]	706.5 (51.5)
Mg (ppm)	234.1 (29.9)	[46.1, 485.5]	244.3 (23.9)	[63.5, 474.8]	239.2 (18.9)
S (ppm)	10.1 (1.2)	[3.4, 29.3]	11.7 (1.3)	[5.3, 30.3]	10.9 (0.9)
Zn (ppm)	22.9 (2.9)	[2.8, 51.2]	20.7 (2.4)	[5.7, 54.0]	21.8 (1.9)

The metals examined were highly correlated with one another and explained the greatest percentage of variability of all other edaphic factors examined in the a priori PC analysis, followed by field capacity.

Zinc shared the highest R^2 in both Chinese tallow and native plots for Axis 1 (0.91 and 0.89, respectively),

and was thus further analyzed as representative of the metals to prevent overfitting of models. Zinc

showed weak correlations in Chinese tallow plots to Chinese tallow QMD and native species density,

with no correlations observed in native plots. There was also a weak positive correlation between

metals and percent visible sky and leaf area index in native plots ($p < 0.10$), but not in Chinese tallow

plots. This reinforces the pattern of weak and inconsistent correlations, at best, between stand

structural or light metrics and edaphic variables assessed in this study. This may be due, in part, to a

lack of variability across our study site as evidenced by the small ranges for many variables in Table 9.

4. Discussion

Consistent with our first hypothesis, areas with greater Chinese tallow abundance had reduced native species density in both the understory and overstory. This reduced native species density was reflected in differences in overstory diversity, although it was not observed in the understory diversity data. In the overstory, Chinese tallow and native oaks shared an inverse relationship in terms of stocking. Where Chinese tallow was established, it was present at high densities of smaller diameter stems. Additionally, those areas had minimal stocking of residual oaks. Areas with little to no Chinese tallow, in contrast, were characterized by larger basal areas of native species, mainly oaks. Our results are consistent with the literature, which generally shows that areas with an increased abundance of invasive species are characterized by a lesser overstory density and diversity, particularly of native species (Hejda *et al.*, 2009; Vilà *et al.*, 2011). However, the lack of difference between plot types for understory diversity differs from some reports in the literature from other forest ecosystems that show reduced understory diversity due to increased propagule pressure and changes in the light environment caused by invasive overstory trees (Reinhart *et al.*, 2005). Because there remains much that we do not know about the relationship between overstory and understory biodiversity (Barbier *et al.*, 2008), it is unclear what the cause may be for similar levels of understory diversity in both Chinese tallow and native plots.

Also consistent with our first hypothesis, Chinese tallow was associated with reduced overall stocking of native forests. This is the first published report we are aware of addressing the correlation between Chinese tallow abundance and stand stocking. It is not clear from this dataset if this reduction in stocking is due to a lower biological carrying capacity of Chinese tallow on these sites, or the more likely explanation that Chinese tallow plots were in areas that were disturbed more recently than the native plots, and thus are in an earlier stage of stand development where stocking is expected to be lower. We did not assess stand ages, and thus cannot directly address this hypothesis. It does appear that mature

oaks may be able to competitively exclude Chinese tallow, but on sites without mature oaks Chinese tallow is able to successfully establish and grow, eventually replacing the oak component. This likely occurred following disturbances that resulted in mortality of mature oak stands based on the amount of downed woody debris observed on many of the Chinese tallow plots (data not shown). Lower QMD and greater density are also consistent with forests earlier in stem exclusion (Oliver and Larson, 1996), as was the observed reduction in trees ha^{-1} from 2012 to 2013. Inter- and intra-specific competition resulting in density dependent mortality may have been a factor. However, there are other confounding factors we did not assess, such as seed source and predation, that could also explain the observed correlations.

Despite the difference in current stand condition, Figure 4 does suggest Chinese tallow plots have the potential to follow similar growth trajectories to native plots. This further supports the inference that differences in stocking may simply be due time since disturbance. Additionally, the high level of stocking in Figure 4 represents the first published maximum SDI recommendations for bottomland hardwoods that we are aware of, and is substantially greater than values derived from the central hardwoods region (Gingrich, 1967; Williams, 2003; Johnson *et al.*, 2009). It should be noted that the geographical region of our study site was very narrow for this maximum SDI to be broadly applicable without further testing.

Contrary to our second hypothesis, there was no correlation between Chinese tallow presence and changes to the below canopy light environment, despite the small difference observed in LAI. While canopy openings could have allowed initial establishment of Chinese tallow, consistent with our hypothesis, it appears that following establishment the invasive species' presence did not alter the light environment compared to native plots based on the metrics we assessed. This is one likely explanation for the lack of difference in understory diversity between Chinese tallow and native plots. While hemispherical canopy photography is suitable to assess the quantity of light penetrating through a

forest canopy, it does not measure the quality (e.g. spectral shifts) of that light. When a canopy gap forms that increases available light in the understory, rapid growth occurs in oak seedling sprouts (Hall and Harcombe, 2001; Lin *et al.*, 2004). However, mesic sites often do not exhibit the same effect due to greater shading by competing vegetation (Lorimer, 1993). Light intensity near the floor of hardwood stands is often at or below the compensation point for oaks. Chinese tallow, a shade-tolerant species, is better suited to grow in lower light conditions, where it can outcompete oaks (Jones and McLeod, 1990; Barrilleaux and Grace, 2000). However, unlike some other shade-tolerant species, studies have shown that its growth rate increases with an increase in light availability (Siemann and Rogers, 2003a). While many small-scale field studies or greenhouse studies have quantified these effects (Rogers and Siemann, 2002; Zou *et al.*, 2009), our data support their conclusions on a larger spatial and longer temporal scale based on the low abundance of oaks in tallow plots we observed. Again, the observed link is indirect, as we did not assess seed source, predation, or other potentially confounding factors.

We were also unable to support our third hypothesis, that Chinese tallow would be more abundant in areas with greater nutrient availability, based on total soil N or plant available nutrient data. While soil acidity and texture did correlate weakly to a handful of differences in the light environment and stand structure of all species or native species, no correlations were observed between Chinese tallow density, basal area, or QMD and most soil variables. Prior research has demonstrated Chinese tallow's high phenotypic plasticity within the southeastern United States (Siemann and Rogers, 2007). This plasticity allows Chinese tallow to establish and grow from upland to bottomland sites under varying hydrological and light conditions (Zou *et al.*, 2009). Beyond a lack of correlations to nutrient availability on these plots, the rusted steel rod technique did not reveal any differences in seasonal surface hydrology across our site. Across all stands, the water table was at or above the soil surface in winter and below 50 cm in depth during summer (data not shown). Thus, it is possible that we were unable to observe many correlations between soil nutrition or hydrology and Chinese tallow's impacts on stand structure due to

a lack of sufficient variability in these independent variables across our study area. This further suggests from a management perspective that newly disturbed areas at this study site will likely regenerate with a greater abundance of tallow than is currently present. However, it is also possible that soil sampling at depths greater than 10 cm and more detailed hydrological measurements might better correlate with some of the dependent variables we tested.

Typical silvicultural systems applied to sweetgum – red oak cover types include clearcut, shelterwood, and group selection regeneration methods (Meadows and Stanturf, 1997), although there are some potential concerns in some areas of the south for the application of shelterwoods and group selection (Oliver *et al.*, 2005; Holladay *et al.*, 2006). If sites are susceptible to Chinese tallow invasion, our data suggest that if Chinese tallow establishes during regeneration, lower stocking, reduced overstory diversity, and a greater density of smaller diameter stems are to be expected during the early portion of the rotation. Management actions that may improve the competitiveness of oak advance reproduction, such as a shelterwood regeneration method, are advisable in east Texas based on the reduced abundance of Chinese tallow in areas with greater oak density. Additionally regeneration methods such as group selection may allow managers to determine if a site is susceptible to Chinese tallow invasion without the risk of converting an entire stand to a Chinese tallow monoculture. However, we recognize that it is unlikely in many situations that these factors will be the overriding influence dictating the selection of the silvicultural system, as compared with other considerations such as site access, operational efficiency, or economic constraints.

Intermediate treatments and monitoring are also key management activities to reduce Chinese tallow abundance (Webster *et al.*, 2006). Intermediate treatments should focus on managing for a healthy native overstory, and thus should follow typical density management recommendations (Goelz and Meadows, 1997; Meadows and Stanturf, 1997). Beyond managed removals, rapid detection of canopy

gaps resulting from other sources of mortality and disturbances would allow for identification of areas that are likely to be susceptible to Chinese tallow invasion.

With aggressive and adaptable species like Chinese tallow, it is important to acknowledge management practices and their value in the field (Miller *et al.*, 2010). The paired-plot design used in this study should serve as an effective means to investigate Chinese tallow's response to the herbicide treatments being applied within bottomland hardwood forests. However, since herbicide treatments began during the second year, there has not been enough time to properly explore the efficacy of herbicides on Chinese tallow recruitment yet on these sites. Given a few more seasons, further data collection will allow for better assessment of various control treatments on stand dynamics.

5. Conclusion

Although there have been several studies focusing on Chinese tallow in bottomland hardwood forests, few studies have been performed evaluating the stand dynamics of mature and aggrading stands. We found that Chinese tallow abundance was correlated to reduced overstory diversity and stand stocking, but was not correlated to an altered below-canopy light environment. Stands with abundant Chinese tallow on our site seemed to be more prevalent in the stem exclusion stage characteristic of earlier successional forests. In our study, low variability in edaphic factors produced weak correlations at best with Chinese tallow stand structure. Chinese tallow is expected to expand its geographic range over the coming years, posing a severe threat to the ecological stability of native forested wetlands. As such, it is crucial to continue to study factors associated with Chinese tallow's ecological success in bottomland hardwood forests.

Acknowledgments

We would like to acknowledge Working Lands Investment Partners, LLC., Stephen F. Austin State University, and the McIntire-Stennis Cooperative Forestry Research Program for providing funding for this study. Hans Williams, Dean Coble, Brian Oswald, Michael Maurer, Warren Conway, Brian Lockhart, and one anonymous reviewer provided insightful reviews that improved this manuscript. Trevor Walker, Wayne Weatherford, Stefni Deaton, Brice Wells, and Michael Felps were instrumental in field data collection and lab work.

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