Overstory management to regenerate longleaf pine and control invasive Chinese tallow in fire ecosystems

by

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Abstract

The native longleaf pine ecosystem is one of the most important, and depleted, ecosystems in the southeastern United States. Restoring this habitat is of major importance to land managers and conservationists alike. Unfortunately, invasive species such as the Chinese tallow tree have become a problem in areas within the natural geographic range of longleaf pine. Invasive species threaten to change the way forests are managed and how restoration efforts are conducted. The purpose of my research was twofold; the first objective was to understand the natural regeneration patterns of longleaf pine as related to different basal area management classes and fire history, as well as evaluating whether there is a clear zone of exclusion around mature trees. The second objective was looking at the re-sprouting characteristics of Chinese tallow following felling operations in association with prescribed burning, canopy closure, stump diameter size, and cutting height. We found that longleaf pine regeneration was successful only at low basal areas (<60 square feet/acre) even though seedlings occurred across a wide range of basal area sites (29-127 square feet/acre). There was little evidence of seedling or sapling exclusion near existing overstory trees. Chinese tallow stump sprouting was not controlled by mechanical cutting or fire except under conditions of high canopy cover. The mortality rate in these areas was 40% compared to just 1% in low canopy cover areas.

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I would like to extend my greatest sincerity to Dr. Fan for bringing me on as a graduate student allowing me to showcase my ability to succeed as a student and a researcher. Without his guidance, completion of my Master's degree would have been especially hard. I am also thankful for Dr. Kush and the ability to learn so much about the longleaf tree from him. His passion for this species is incredible and contagious. I also want to thank Dr. Lauer for providing valuable input on my statistics and comments throughout my thesis. I would also like to thank Shaoyang Yang for the continued support in and out of the field. Without him, this research would not have been possible. I am also grateful to Tyler Baxter, Stathon Wilson, Dan Gill and Sunil Nepal for helping me establish the plots and for assistance with data collection in the field. Lastly, I thank the Mississippi Sandhill Crane National Wildlife Refuge staff for their assistance in providing us access to the refuge, in addition to giving us advice and updated maps.

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List of Abbreviations

DBH	Diameter of a tree at breast height (4.5’)
GLD	Ground Line Diameter
MSCNWR	Mississippi Sandhill Crane National Wildlife Refuge
GBNERR	Grand Bay National Estuarine Research Reserve
RLGS	Regional Longleaf Pine Growth Study
EEF	Escambia Experimental Forest
QMD	Quadratic Mean Diameter
TPA	Trees per Acre
ASC	Average Sprouting Capacity

Chapter 1

Effects of prescribed fire, canopy closure, stump diameter, tree age and cutting height on Chinese tallow (*Triadica sebifera*) stump sprouting

1.1 Introduction

Invasive species impact native plant and animal communities alike. Community dominance, alteration of native ecosystem processes (Gordon, 1998), and major economic losses (Gifford and Armacost, 2012) are just some of the effects invasive plants pose in an ecosystem. Invasives have the ability to change the physical structure of habitats and alter the usability of those habitats for other native species (Gifford and Armacost, 2012). Because of this, invasion by exotic plants into forested ecosystems has become a major concern, recognized by ecologists, conservationists, and land managers worldwide (Gordon, 1998; Camarillo et al., 2015).

Native to central China (Burns and Miller, 2004), *Triadica sebifera* (L.) Small (hereafter Chinese tallow) was originally introduced to the United States in the 1700's as a crop species (Bruce et al., 1997) and has since been aggressively invading ecosystems; jeopardizing many important native ecological functions where it is found (Bennett et al., 2015). Chinese tallow is an invasive species commonly found in the southeastern United States (Figure 1.1), ranging from Texas to Florida and up to North Carolina (Conway et al., 2002; Adams and Saenz, 2012; Bennett et al., 2015). The northern extent of its existence is northern Alabama and Mississippi with a few localized populations in Tennessee and Kentucky (EDDMapS, 2017). Chinese tallow is also found in California where it just began producing naturalizing populations (Bower et al., 2009). What makes this tree such an issue is its ability to form monotypic stands (Gifford and

Armacost, 2012) due to its rapid growth rates, large annual seed loads (Conway and Smith, 2002), high fecundity, and the fact that it has few pests or pathogens (Wang et al., 2014). Chinese tallow also has high shade tolerance giving it an ability to grow in understories and respond to release when bigger nearby trees fall or are harvested (Miller, 2003). Although the Chinese tallow tree has a relatively short local seed dispersal distance (distance the seed falls from the seed tree), it has an effective dispersal distance of nearly 28 km that is influenced by bird dispersal and/or being carried by rivers, streams, and storm runoff with few limiting barriers (Fan et al., 2012). According to Wang et al. (2014), if current management of Chinese tallow continues, expansion of Chinese tallow into uninvaded areas is likely to continue. Because of this, Chinese tallow continues to be a growing concern.

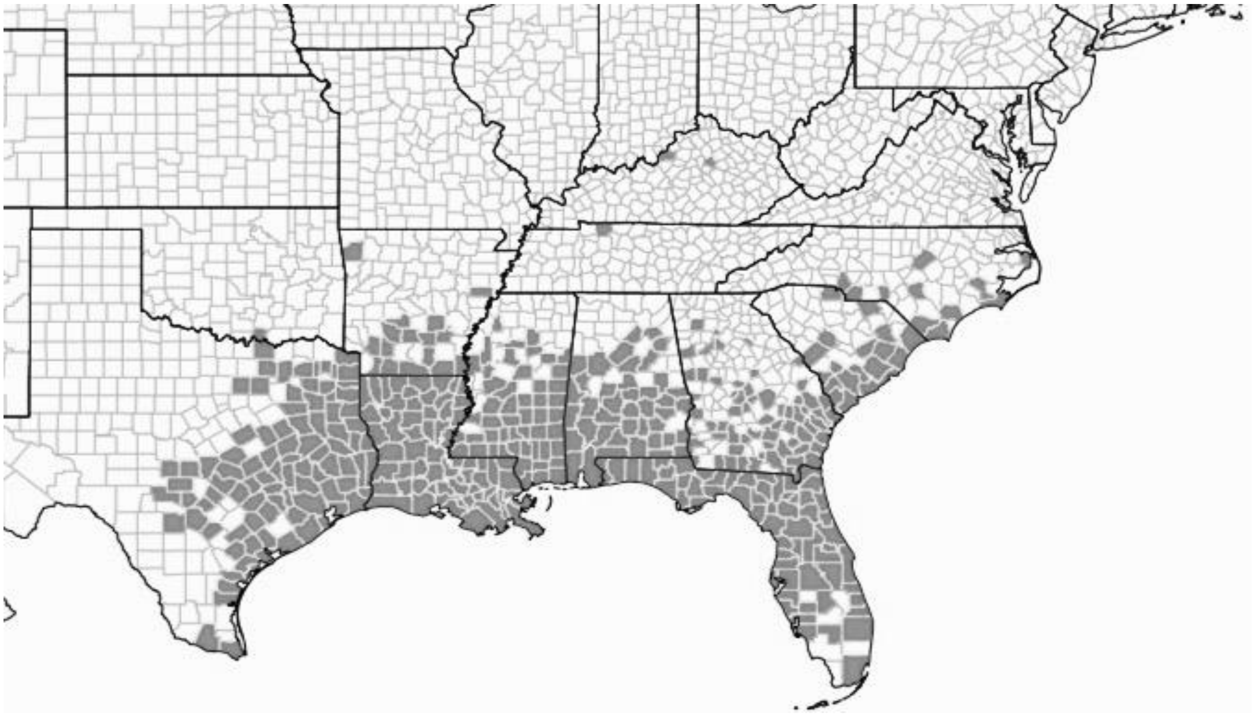


Figure 1.1. The current reported range of Chinese tallow in the southeastern United States. Photo derived from EDDMapS, 2017.

Current management of Chinese tallow is focused on a variety of different techniques. Ansari and Nand (1987) reported that types of cattle such as goats and sheep will eat tallow leaves. However, due to the toxicity in its leaves, grazing is rarely used as an effective approach to control the tree (Russell et al., 1969). Another method of control, though rarely plausible to use, is through mechanical removal with the use of heavy equipment (Bruce et al., 1997). This is not a suitable way to control Chinese tallow because this disturbs native soils and non-target vegetation (Bruce et al., 1997) making it easier for other non-native species to colonize. Methods easier on the local environment include hand removal of individual saplings <1 m tall (Bruce et al., 1997). However, it is unlikely this is a feasible method if you have thousands of seedlings/saplings per acre. It has been reported that prescribed burning controls Chinese tallow <2 m tall (Burns et al., 1997). One of the most common methods of control is through the use of herbicide. Whether it consists of cutting the tree down and then applying an herbicide on the stump in order to prevent stump sprouting, or applying herbicide through the bark, these two techniques are the most effective use of control (Bruce et al., 1997). Many landowners are likely to try cutting Chinese tallow down as a method of control, but will soon find out that it is likely to re-sprout.

Many trees depend on stump sprouting as a form of regeneration (Keyser and Zarnoch, 2014). Sprouts from highly developed root systems of the parent tree exhibit rapid growth and can impact post-disturbance species composition (Keyser and Zarnoch, 2014). However, stump sprouting in an invasive species such as Chinese tallow can be largely unwanted, and usually requires the use of herbicide to stop it (USDA Plant Guide). Despite Chinese tallow’s known ability for producing stump sprouts, there are surprisingly no quantitative studies that have investigated the ability to sprout across a range of factors. Thus, we know little about which factors may help reduce the ability of this invasive species to re-sprout following felling.

Tree age, stump diameter (Ducrey and Turrell, 1992; Randall et al., 2005; Wu et al., 2008), stump height (Jobidon, 1997), prescribed fire, and canopy closure can all have an effect on stump sprouting. Some of the above factors such as tree age and diameter cannot be controlled and vary among individuals, whereas stump height, prescribed fire, and canopy closure can be controlled. Chinese tallow may have a different re-sprouting response associated with these factors. Examples of other species varying their response is the genus *Quercus* (oak) where research has shown that as oak stump diameter increased, the number of sprouts per stump decreased (Khan and Tripathi, 1986). That is in contrast to Wu et al. (2008) that showed in 62 species, the number of sprouts per stump increased while the stump diameter increased.

Our study investigated the re-sprouting of Chinese tallow tree following felling to identify situations where the re-sprouting ability is minimized or controlled completely. The first hypothesis we tested was that there is no difference in re-sprouting ability between the different factors such as age, diameter, etc. The second hypothesis we tested is that there is no effective way to control the re-sprouting without the use of any herbicides. Specifically, we hypothesized that 1) tree age, stump diameter, cutting height on a tree (stump height), fire, and/or canopy closure will have some effect on the number and size of the sprouts, and 2) there will be a situation where Chinese tallow can be controlled without the use of herbicides. These results may help land managers identify effective management techniques to improve control of this invasive species.

1.2 Materials and methods

1.2.1 Study site

This study was carried out in the Southeastern Coastal Plain of Mississippi, at the Mississippi Sandhill Crane National Wildlife Refuge (MSCNWR; 30°27’N – 88°39’W) and the Grand Bay National Estuarine Research Reserve (GBNERR; 30°25’N – 88°25’W; Figure 1.2). Both sites belong near the center of its current range (Figure 1.2). Having our study site in Mississippi where Chinese tallow has increased in abundance by 445% from 1994 to 2006 (Oswalt, 2010) gives us a unique opportunity to examine a vast array of factors that influence stump sprouting. The area is characterized by a warm temperate climate with poorly drained soils (Norquist, 1984) and a low (average elevation = 5 m), flat topography. The MSCNWR and GBNERR encompass a mosaic of pine savannas and pinelands interspersed with forested swamps, drainages, and estuarine marshes.

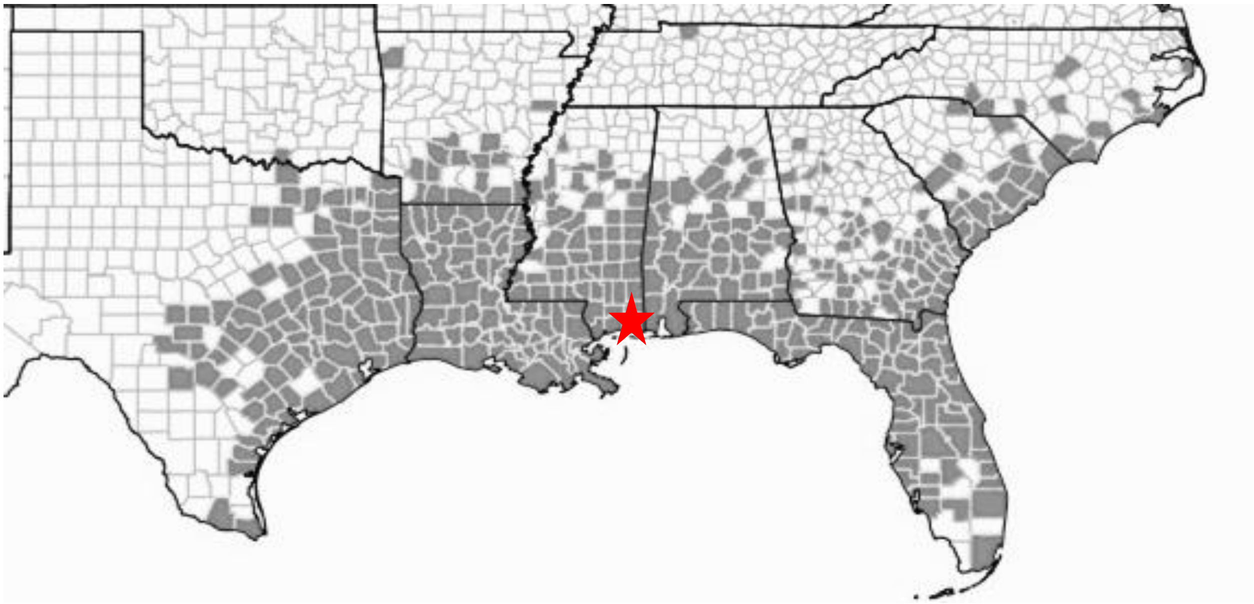


Figure 1.2. A map of the southeastern United States, with the red star indicating the location of the two study sites in south Mississippi. The shaded dark grey area is where Chinese tallow is currently reported.

Current management of the two study sites focuses on restoring the wet pine savanna through the use of prescribed fire to promote nesting habitat of the Mississippi Sandhill crane. Prescribed fire is implemented in the spring and fall to clear woody vegetation, improve areas overstocked with slash pine (*Pinus elliottii*), and enhance the suitability of nesting areas of the cranes. Since 1985, the MSCNWR has been conducting prescribed fires on 1,000-9,000 acres per year, with each stand on a one- to five-year fire rotation. Large proportions of the area have various stocking levels of longleaf pine (*Pinus palustris* Mill.) and are variably infested by invasive species (e.g., Chinese tallow, cogon grass (*Imperata cylindrica* L.), Japanese climbing fern (*Lygodium japonicum* (Thunb. Ex. Murr.) Sw.), etc.).

1.2.2 Study design

We looked specifically at what effects age, tree diameter at breast height (DBH), cutting height of the stump, stump ground line diameter (GLD), prescribed burning (pre-felling), and canopy closure have on sprouting ability. We had three different study locations (two in the MSCNWR, one in the GBNERR) where plots were created with different treatments applied to each site. Location one in the MSCNWR served as our low canopy closure site with different cutting heights applied to trees. Location two in the MSCNWR served as our medium canopy closure site and our site with a prescribed fire implemented two weeks prior to felling. Location three (in the GBNERR) was sub-divided into three plots which served as high, medium, and low canopy closure sites.

Location one consisted of one permanent plot (approximately 30m X 43m) which was established in March, 2016. This plot was divided into five sub-plots with each receiving a different cutting height treatment (Figure 1.3).

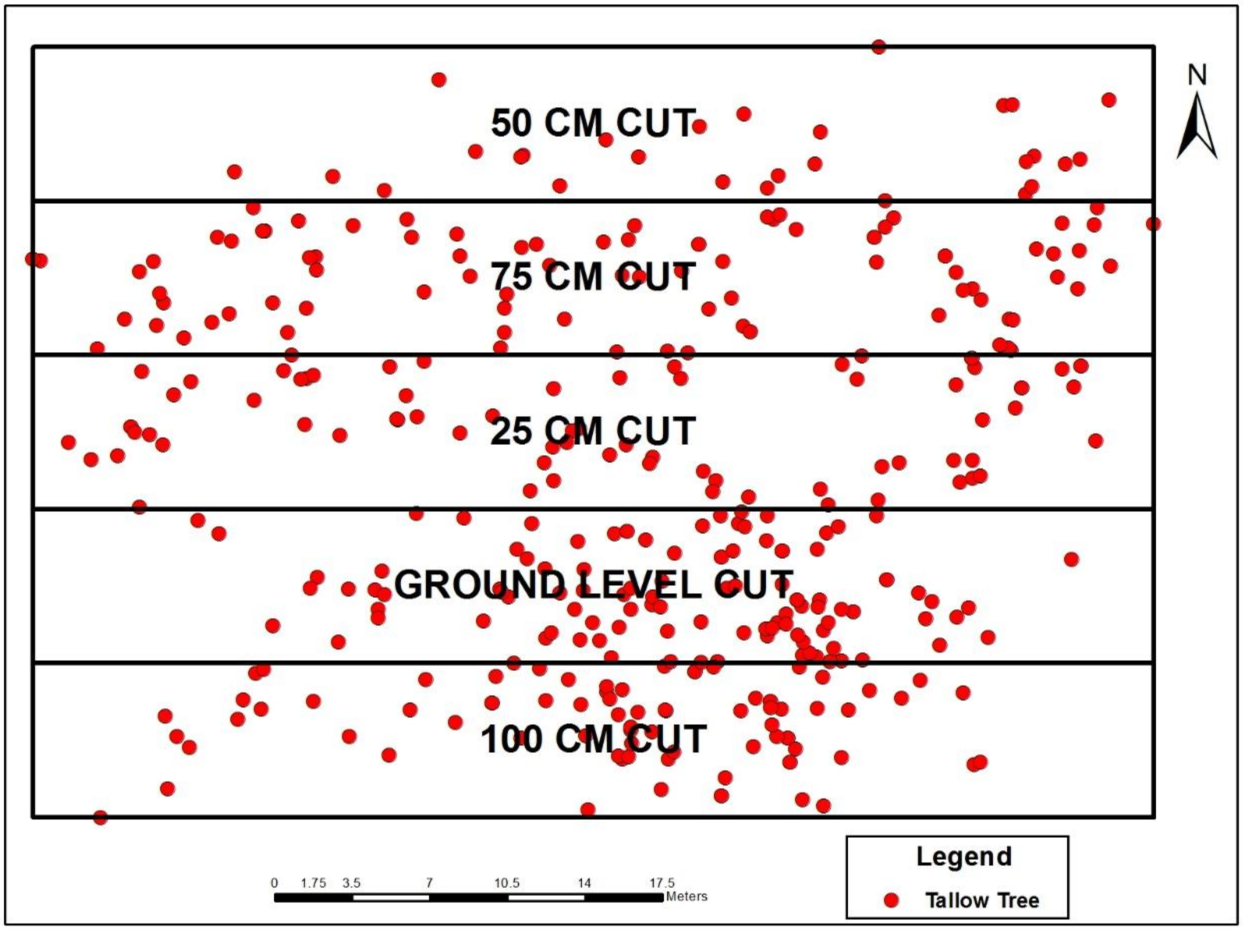


Figure 1.3. The layout of location one plot in the Mississippi Sandhill Crane National Wildlife Refuge created March, 2016. The total size of the plot was approximately 30 m X 43 m, with each sub-plot approximately 6.1 m X 43 m. Treatments consisted of different cutting heights (cm) above the ground.

There were 5 cutting height treatments consisting of ground-level, 25, 50, 75, and 100 cm applied at random to strips within the main plot. At the time of plot establishment, every Chinese tallow tree present was cut at the indicated height. In total, 378 trees were felled and the following data were recorded: age, total height, DBH, GLD, stump height, and whether fire had burned around the base of the tree. A prescribed fire was implemented in this burn unit two weeks prior to us cutting the trees and burned a portion of the area. The burned portion of each treatment plot was easily identified by char and nearby vegetation being brown (burnt) or green (not burnt). There were a few non-target species left in the stand such as slash pine, longleaf pine, and bald cypress (*Taxodium distichum* (L.) Rich). Since this plot was to serve as our low canopy closure treatment, no efforts were used to remove the remaining trees. This plot had an average canopy closure of 34.0% (standard deviation = 31.6). All canopy measurements were made using a densitometer. The basal area of Chinese tallow in this plot was 54.0 ft^2/acre with 1,186 trees/acre (TPA).

Location two consisted of one permanent plot (approximately 55m X 112m) which was established in May, 2016 (Figure 1.4). This plot was not intended to serve as a test for stump height, therefore, all cuts were made at a cutting height ≤50 cm above ground. The same data were recorded as in the first plot. There had been a prescribed fire two weeks before cutting that affected every single stem. In total, 130 trees were felled in this plot. Canopy closure (measured using a densitometer) for this plot averaged 52.0% (standard deviation = 22.9). The basal area of Chinese tallow in this plot was 7.2 ft^2/acre with 85 TPA.

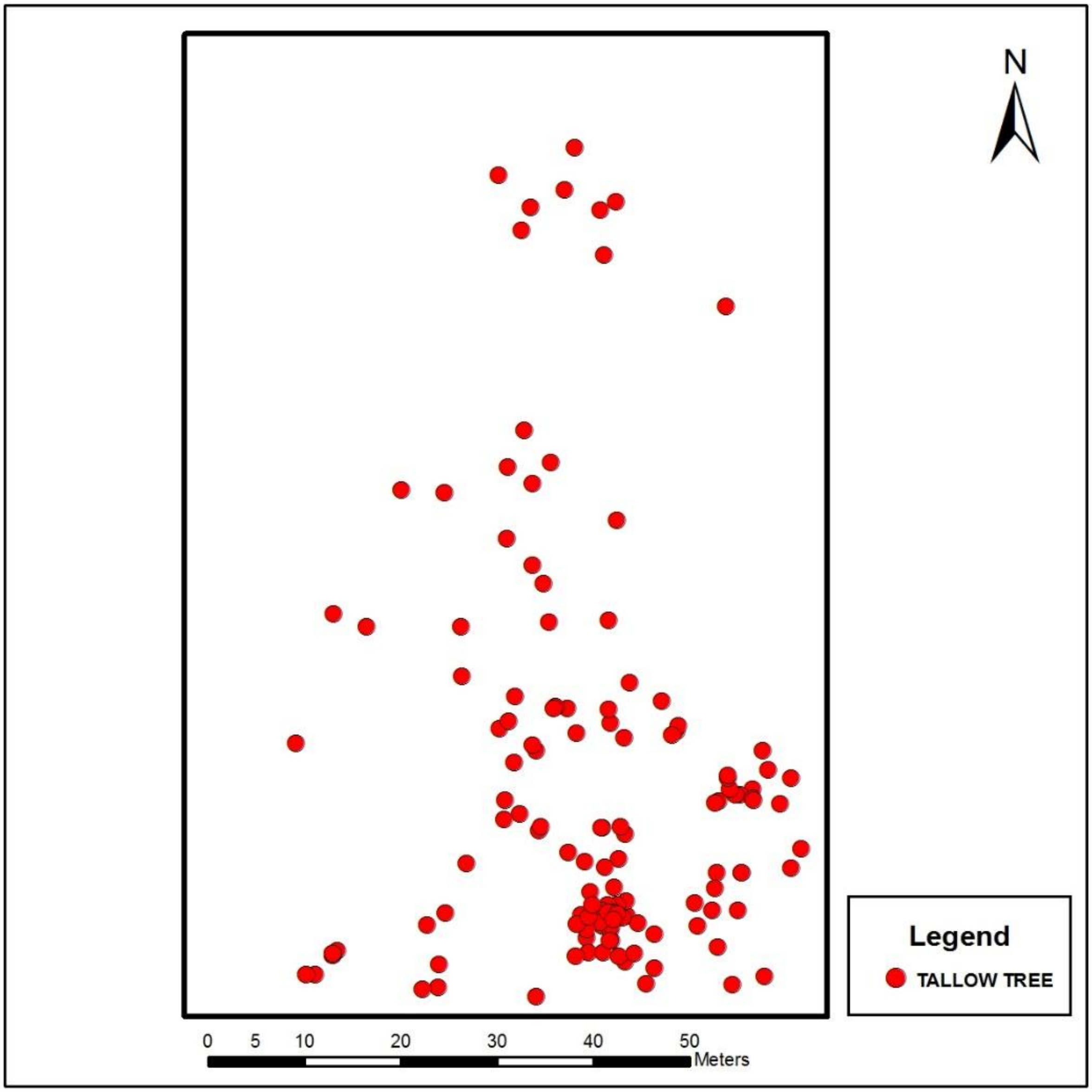


Figure 1.4. The layout of location two plot in the Mississippi Sandhill Crane National Wildlife Refuge created in May, 2016. Canopy closure was 52% for the plot. A prescribed fire was conducted two weeks prior to cutting our trees.

Location three consisted of one permanent plot in the GBNERR (approximately 27m x 40m) and was established in May, 2016. This plot was subdivided into three different sub-plots (Figure 1.5), representing different canopy classes (low: average = 39.2%, standard deviation =

24.2; medium: average = 50.0%, standard deviation = 18.7; high: average = 82.5%, standard deviation = 10.8). Canopy measurements were made using a densitometer.

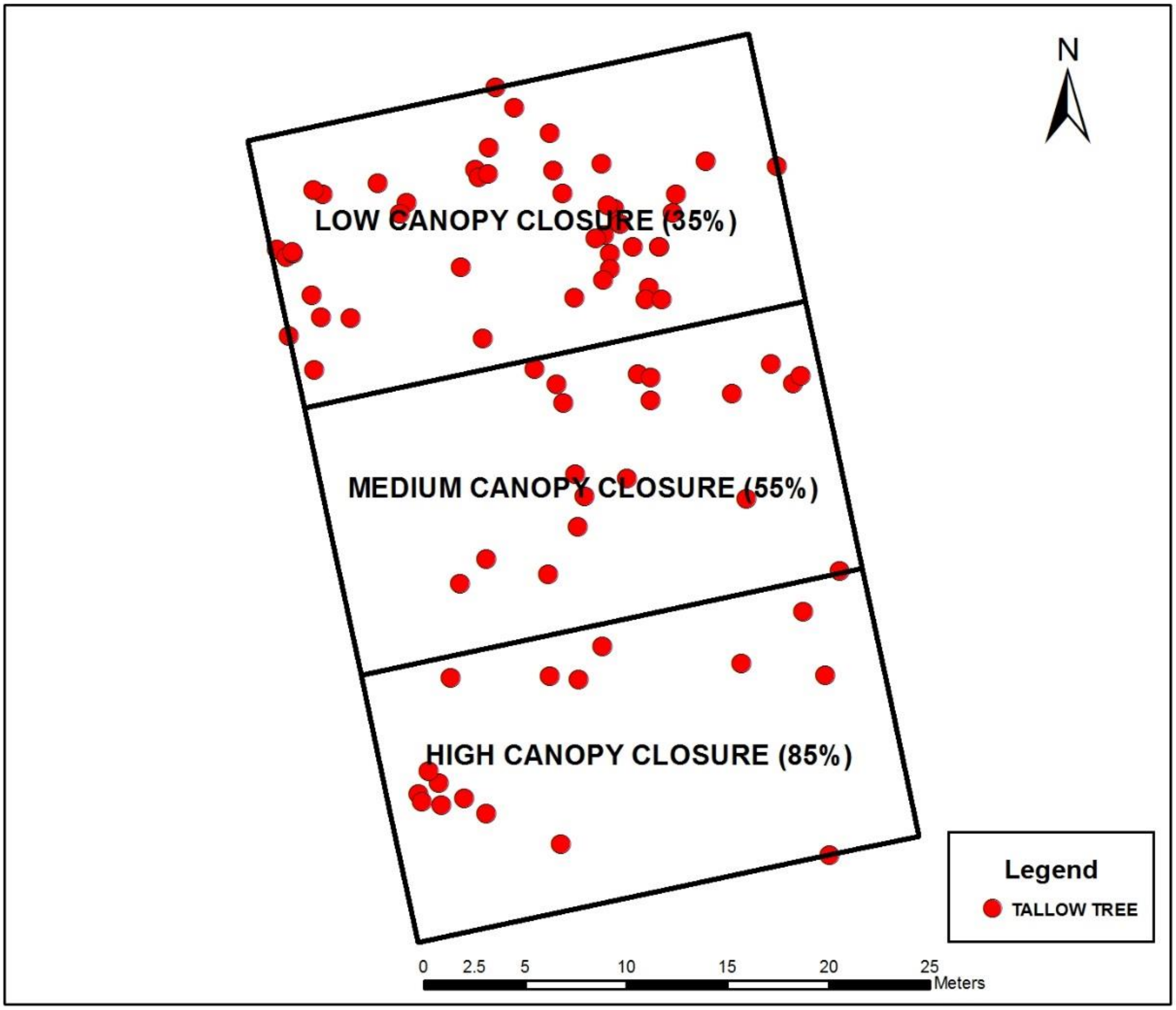


Figure 1.5. The layout of location three plot in the Grand Bay National Estuarine Research Reserve created in May, 2016. Treatments consisted of low, medium, and high canopy closures.

The overstory consisted primarily of mature slash pine. Trees of both species (Chinese tallow and slash pine) were removed to get the canopy closure to the desired state. The same data were recorded as in the other plots. All cuts were made at an easy cutting height (≤50cm above ground). In total, 80 trees were felled in this plot. The basal area and TPA for each sub-plot of location three are summarized in Table 1.1.

Table 1.1. Basal area and trees/acre comparison within each sub plot in the GBNERR.

Plot	Basal area (ft.^{2} / acre)	Trees per acre
Low canopy closure	10.3	337
Medium canopy closure	27.6	337
High canopy closure	9.2	225

For all plots, all trees were cut using a chainsaw. After cutting, a metal tag was fixated at the base of each stump to allow for future identification. Stump characteristics for the three plots and all sub-plots are summarized in Table 1.2.

Table 1.2. Characteristics of Chinese tallow stumps cut in each plot and sub-plot in 2016.

				Stump Diameter Size (cm) and Total Number				
Plot	Cutting Height	Average Canopy Closure (%)	Did fire affect trees?	0-5	5.1-10	10.1-15	15.1-20	> 20
1	Ground Level	34	Some trees	37	17	6	5	1
1	25cm	24	Some trees	30	1	8	7	0
1	50cm	23	Some trees	49	5	4	2	2
1	75cm	25	Some trees	45	5	7	5	2
1	100cm	37	Some trees	90	16	6	0	2
2	≤50cm	52	Yes	27	45	38	8	1
3	≤50cm	39	No	13	15	1	0	0
3	≤50cm	50	No	11	10	6	2	1
3	≤50cm	82	No	7	13	0	0	0

1.2.3 Sprouting measurements

A survey for location one in the MSCNWR occurred 2, 5, 8, and 12 months after treatment (MAT). Location two in the MSCNWR was measured 6 and 10 MAT. The location three plot at GBNERR was measured 3, 6, and 10 MAT. The number of living sprouts, the total aggregated height of all sprouts (to the nearest cm), and the height of the tallest sprout (to the nearest cm) was recorded. Total aggregated height is a common measurement which sums the height of all stems on a single stump and incorporates growth, stunting, dieback, and mortality (Quicke et al. 1996). This measurement was used similarly as Quicke et al. (1996), except they used it on an acre basis whereas our focus was per rootstock. Our observations in the field since 2012 indicate that when a fire runs through the area prior to felling, some stumps appear dead with sprouts forming on lateral roots. Therefore, the presence of lateral root sprout formation was counted similarly as if the sprouts were coming directly from the stump.

1.2.4 Average sprouting capacity

Since sprouts originated from both the top of the stump (where it was cut) and the sides of the stump, a new response variable was created to account for the cylindrical orientation of possible sprout formation (Figure 1.6). The average sprouting capacity (ASC) of an individual stump based off stump height considers this by using the following formula:

$$Stump\ Height\ ASC = \frac{\#\ Sprouts\ 8\ MAT}{\pi * GLD * (Cutting\ Height + 50cm)}$$

The equation is derived from finding the surface area of a cylinder/tube (π * diameter * length). Cutting height + 50 cm is used as the length in the equation since we had stumps that were cut at ground level. This allows us to use the 0 cm cutting height in our analysis; otherwise, the denominator would be a ‘zero’ and the data would not be useful. We used the 8 MAT measurement of total number of sprouts per stump as the numerator. The end result is the number of sprouts on a stump with a certain area covered.

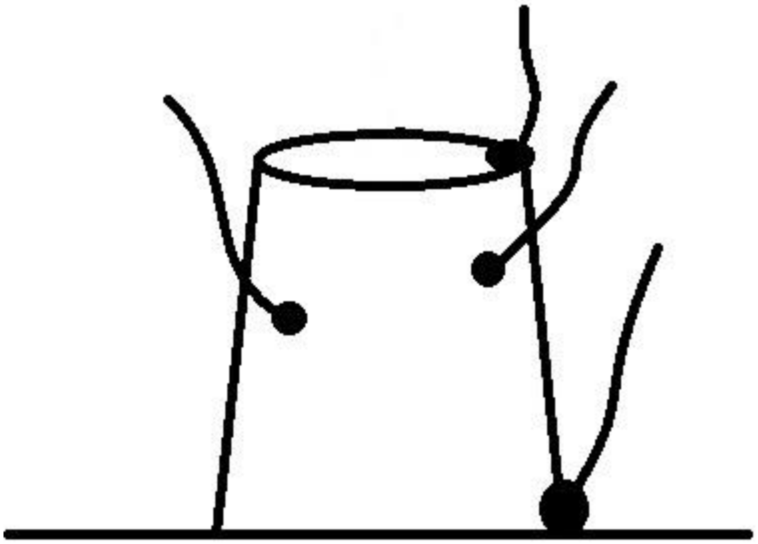


Figure 1.6. Stump height average sprouting capacity (ASC) takes into account the cylindrical orientation of sprout orientation.

Another equation we will use is the ASC disregarding stump height with more focus on the biomass of the root system (Figure 1.7). The formula used to determine is as follows:

$$Root\ ASC = \frac{\#\ Sprouts\ 8\ MAT}{\pi * \left(\frac{GLD}{2}\right)^2}$$

This equation is derived from finding the surface area of a circle (stump) with no interest in the height of the stump. Since this is a measure of sprouts per unit of groundline basal area, it allows us to look at sprout formation with respect to translocation of cabohydrate reserves from the root system.

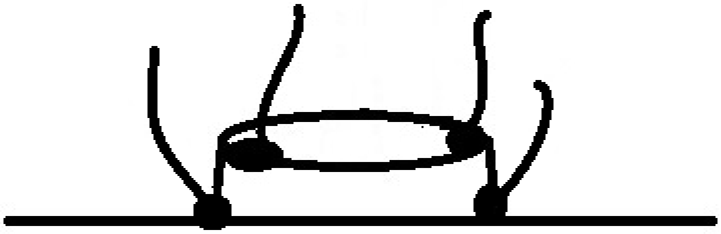


Figure 1.7. Root ASC disregards stump height and is a measure of sprouts per unit of groundline basal area. This takes into account the root biomass and its impact on the sprout potential.

1.2.5 Data analyses

Simple linear regressions were conducted to explore the trends between total aggregated height and total number of sprouts per stump by age and DBH. We then graphed total aggregated height and total number of sprouts per stump by canopy closure. The log-log transform (for normality) of root ASC was used to evaluate the impact of canopy closure and stump diameter. An ANCOVA was then performed using DBH and grouped canopy closure percentage as covariates. Tukey's post-hoc ANCOVA comparisons were then ran to evaluate the significance between the three canopy closure classes. The same method was conducted testing total aggregated height by DBH and grouped canopy closure. A MACOVA was then ran to test the significance of cutting height by the number of sprouts per stump, total aggregated height, and stump height ASC. To evaluate the impact of prescribed fire on total number of sprouts and total aggregated height, stumps were broken into three DBH classes and two sample t-tests were run. All analyses were done using Minitab 17 Statistical Software (Minitab 2010) and R with an alpha level of 0.05 used to determine significance.

1.3 Results

A total of 550 stumps were cut and measured in May, August, and November, 2016 and March, 2017. Tree age revealed the parent trees originating around 1998 (19 years old). The youngest trees were two years old, with a mean age of 7.3 (standard deviation = 3.8). Stump diameters (at the ground level) ranged from 7.6 to 45.2 cm.

1.3.1 Exploratory data analysis

The exploratory data analysis used data from the 8 MAT measurement. Data from location one at the MSCNWR (stump heights <50cm), as well as all data from location two within the MSCNWR and location three at the GBNERR were used to determine possible trends with the data. There was a total of 311 observations, with the results of the total aggregated height on each stump and the total number of sprouts vs. age and DBH shown in Figure 1.8 (a-d).

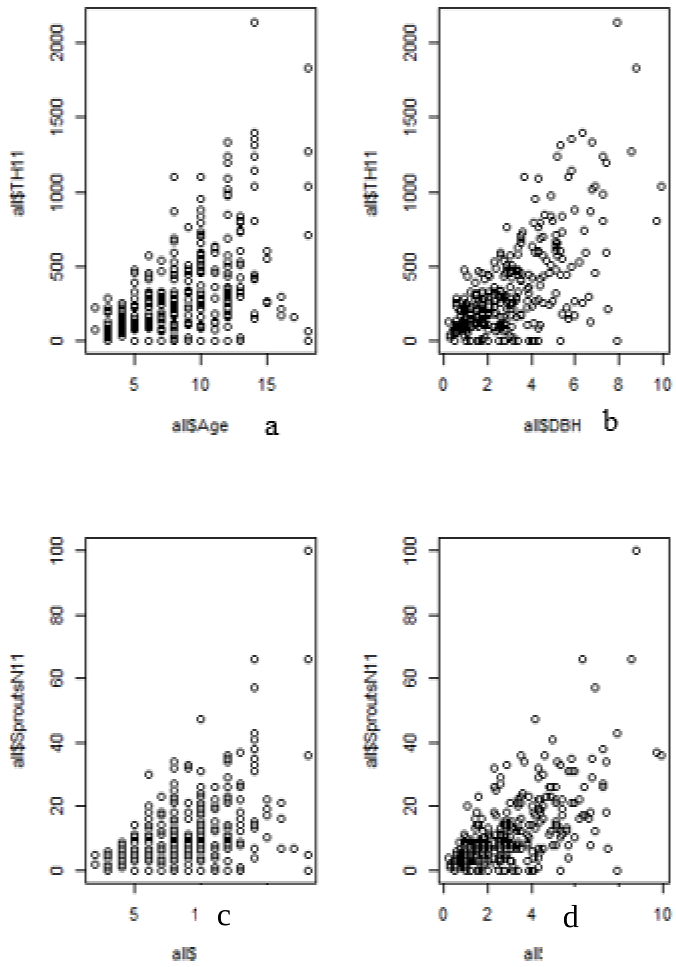


Figure 1.8. The top two graphs (a-b) show total aggregated height on each individual stump vs. age and DBH, respectively. The bottom two graphs (c-d) show the total number of sprouts vs. age and DBH, respectively. All trends appear to have a positive association.

Location one at the MSCNWR had 210 canopy closure measurements taken, with an average canopy closure of 34% throughout the plot. Similarly, the location two plot at the MSCNWR had 66 total canopy measurements taken, with an average closure throughout the plot of 52%. Lastly, each sub-plot at the GBNERR location three plot had six canopy measurements taken. The ‘low’ canopy closure plot had a closure of 39%. The ‘middle’ canopy closure plot had a closure of 50%, whereas the ‘high’ canopy closure plot had a closure of 82%. The total aggregated height (Figure 1.9a) and the total number of sprouts (Figure 1.9b) versus canopy closure is shown in Figure 1.9. The five different plots were classified into three groupings for canopy closure (low, medium, high).

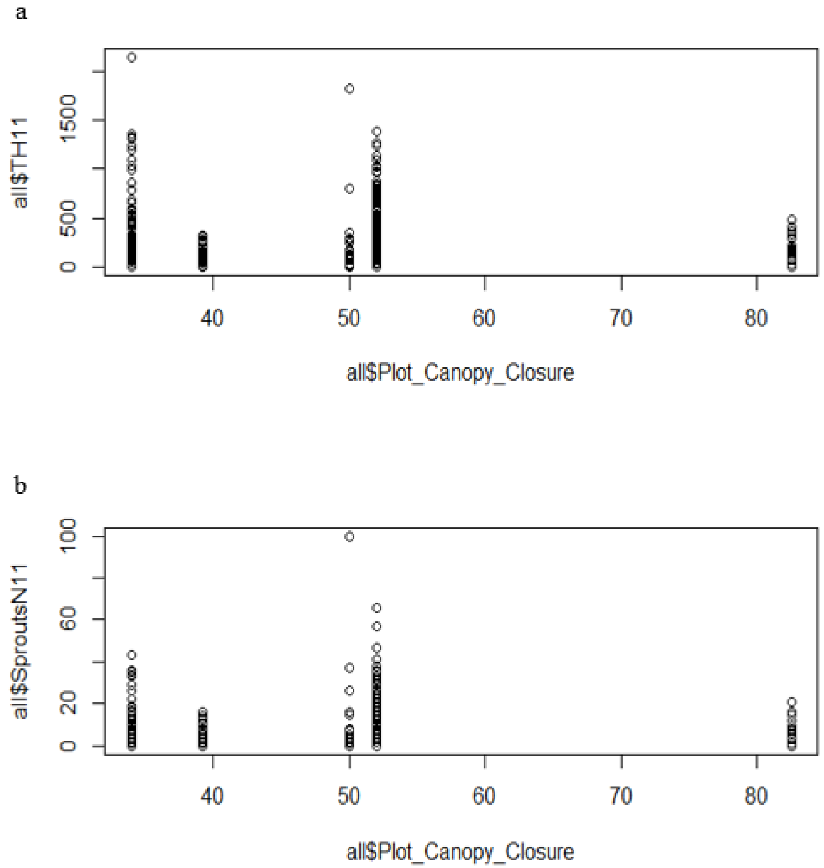
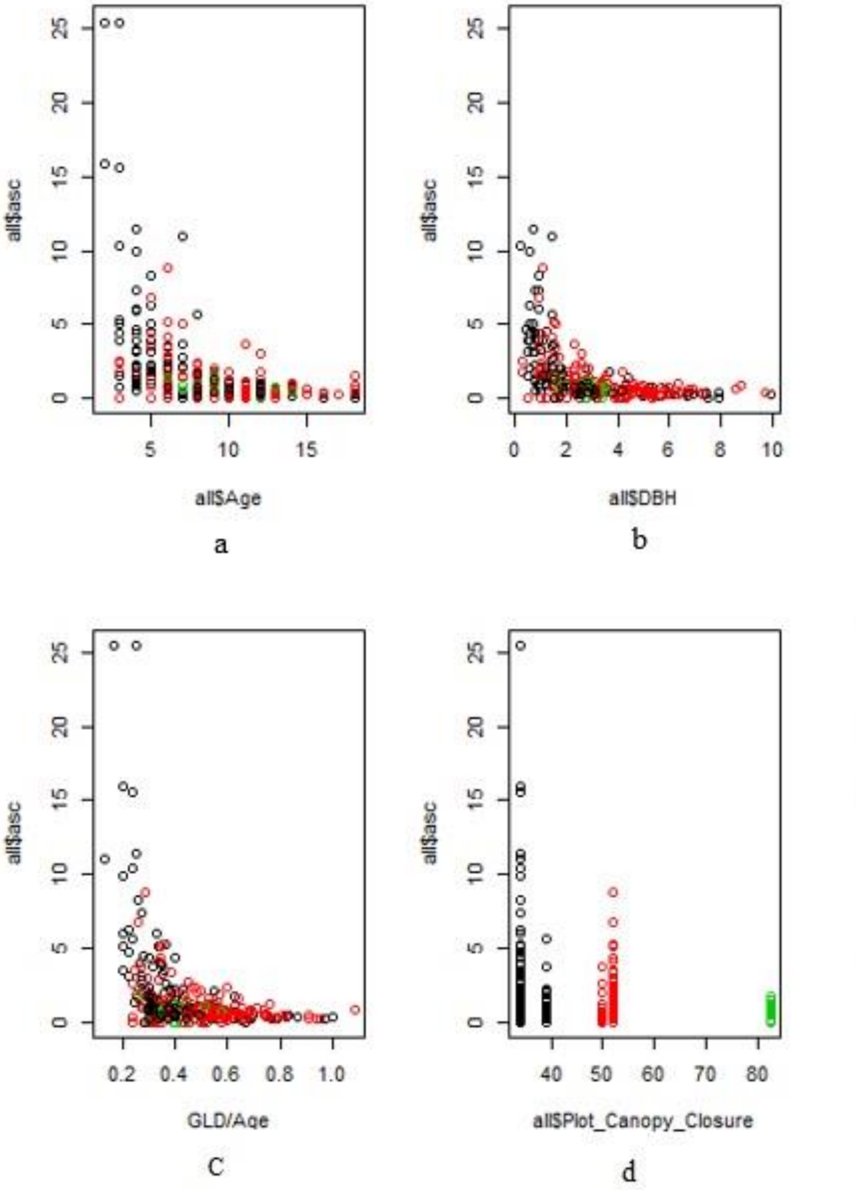


Figure 1.9. a) The total aggregated height of each stump versus plot canopy closure is shown. b) The total number of sprouts versus plot canopy closure is shown.

The last trends we wanted to investigate required transforming the response variable into the root ASC. Figure 1.10 shows these trends. It appears a negative relation occurs with all variables.



a b

c d

Figure 1.10. Root ASC versus age (a), DBH (b), GLD/Age (Annual growth rate; c), and plot percent canopy closure (d).

1.3.2 Impact of canopy closure and stump diameter

The root ASC data was transformed using log-log (for normality) and an ANCOVA was performed using DBH and grouped canopy closure as covariates. We were able to use the log-log transformation with the exclusion of only two ‘zeros’ in our data which were removed and not used in analysis. Both canopy closure ($F_{[2,299]} = 18.97$, $P = < 0.001$) and DBH ($F_{[1,299]} = 154.05$, $P = < 0.001$) were significant in affecting the root ASC log-log (Table 1.3). Tukey’s post-hoc ANCOVA comparisons between the grouped canopy closure and root ASC log-log showed a negative relationship with all three levels of canopy closure differing from one another (Figure 1.11).

Table 1.3. Results of the ANCOVA indicate both canopy closure and DBH significantly affect the transformed root ASC of Chinese tallow.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Canopy Closure	2	2.872	1.436	18.97	<0.001
DBH	1	11.658	11.658	154.05	<0.001
Residuals	299	22.628	0.76		

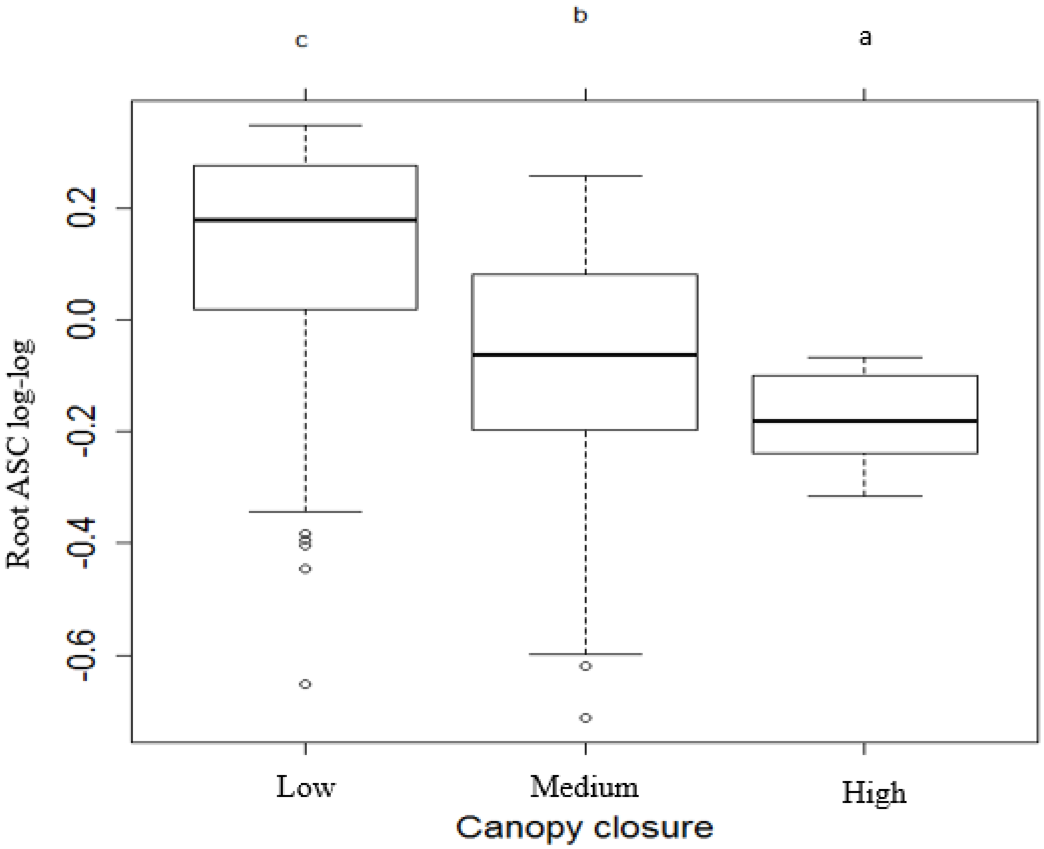


Figure 1.11. Tukey's post-hoc ANCOVA comparisons between canopy closure and root ASC log-log data indicate a negative relationship. All three grouped canopy closures differ in root ASC log-log from one another (2-1 $P = 0.0386$; 3-1 $P = <0.001$; 3-2 $P = 0.0102$).

In addition to using the transformed root ASC log-log data, we also used the untransformed total aggregated height for each stump, and ran an ANCOVA and Tukey's post-hoc comparisons. Similarly to the root ASC log-log data, both canopy closure and DBH had a significant effect on the total height combined of each stump ($F_{[2,299]} = 10.88$, $P < 0.001$; $F_{[1,299]} = 216.61$, $P < 0.001$, respectively; Table 1.4). DBH had a positive association with the total aggregated height as well as the total number of sprouts per stump (Figure 1.12). Running Tukey's post-hoc ANCOVA comparisons on canopy closure indicate that low and medium canopy closure had a significantly greater total aggregated height on an individual stump than did the high canopy class (Figure 1.13).

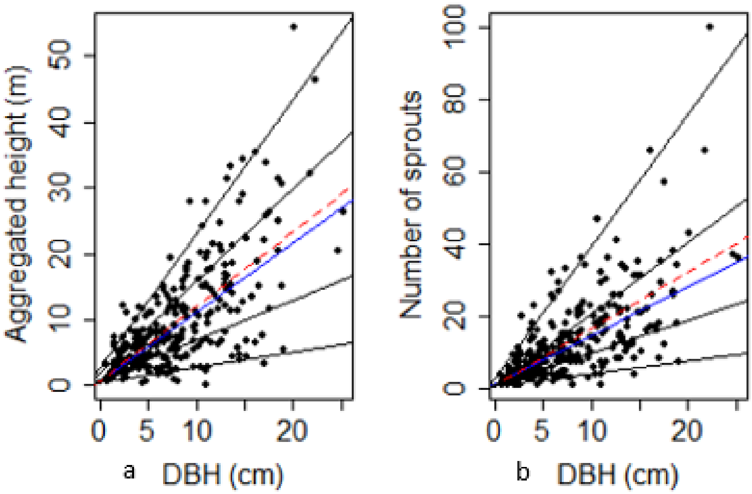


Figure 1.12. Quantile regression to predict the 95%, 75%, mean (red), 50% (blue), 25%, and 5% regression lines based on aggregated height (a) and number of sprouts per stump (b).

Table 1.4. Results of the ANCOVA indicate both canopy closure and DBH significantly affect the total aggregated height of Chinese tallow.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Canopy Closure	2	1289927	644963	10.88	<0.001
DBH	1	12840190	12840190	216.61	<0.001
Residuals	299	17724408	59279		

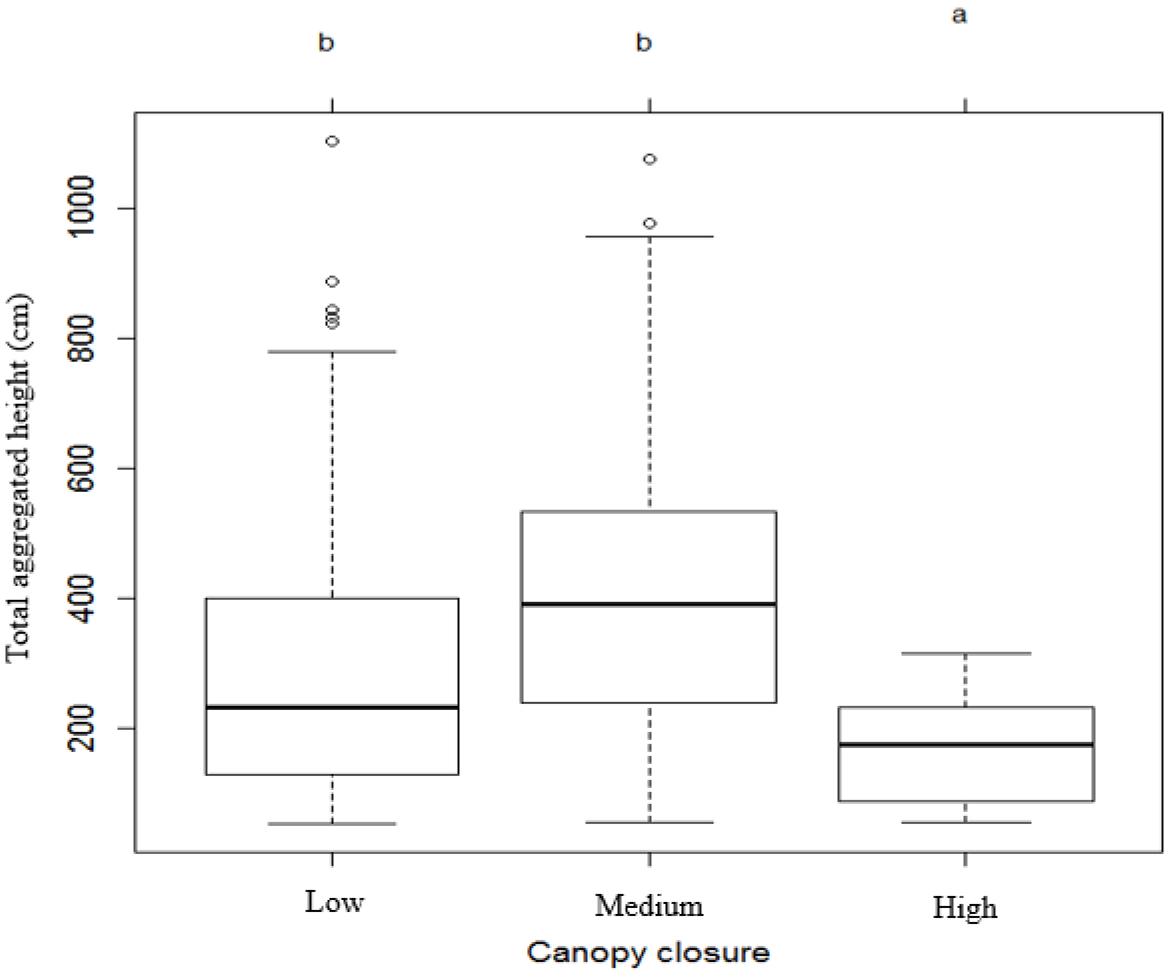


Figure 1.13. Tukey's post-hoc ANCOVA comparisons (at p=0.05) between canopy closure and total aggregated height on an individual stump. Low and medium canopy closure did not differ from one another, whereas high canopy closure was significantly lower than the other two.

1.3.3 Stump mortality

Of the 550 measured stumps, 23 (4.1%) exhibited no sprouting activity. We ran a logistic regression to test if DBH and/or canopy closure had a significant effect on the mortality of stumps. Plot canopy closure was significant (*P* = 0.0002) on stump mortality; however, DBH was not (*P* = 0.499). Stump mortality from each plot is summarized in Table 1.5.

Table 1.5. Stump mortality from all plots in MSCNWR and GBNERR.

Plot	Cutting Height	Average Canopy Closure (%)	Did fire affect trees?	Mortality rate (%)
1	Ground Level	34	Some trees	0%
1	25cm	24	Some trees	2.2%
1	50cm	23	Some trees	0%
1	75cm	25	Some trees	1.6%
1	100cm	37	Some trees	0%
2	≤50cm	52	Yes	2.7%
3	≤50cm	39	No	17.2%
3	≤50cm	50	No	43.3%
3	≤50cm	82	No	40%

1.3.4 Impact of age

Quantile regressions were conducted to determine the relationship between the total aggregated height and the number of sprouts per stump versus the tree age (Figure 1.14; a-b). Additionally, root ASC had a non-linear decline with increased age (Figure 1.14c).

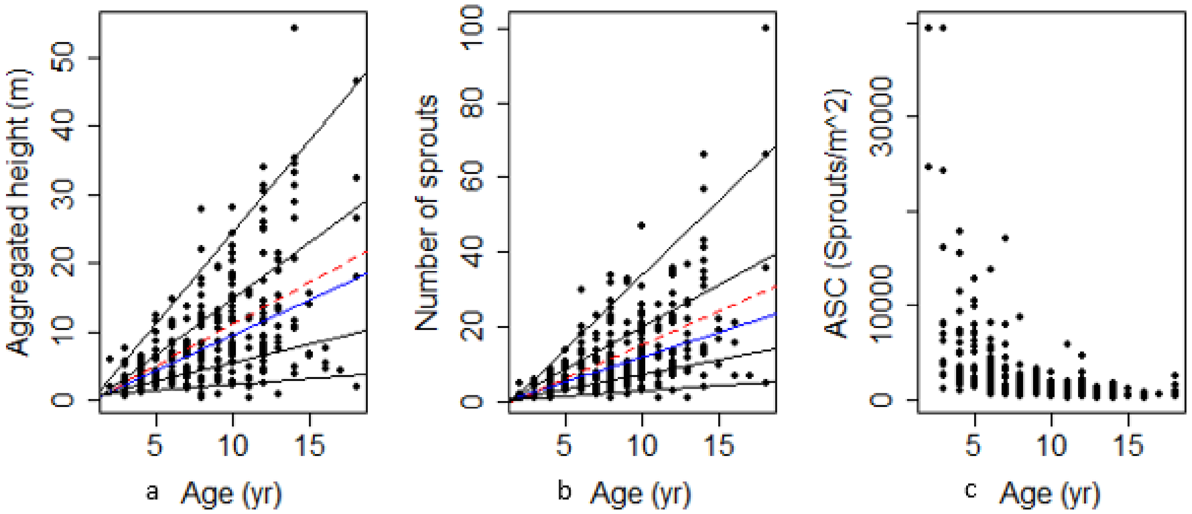


Figure 1.14. Quantile regression were used to predict the 95%, 75%, 50% (blue), mean (red), 25%, and 5% regression lines based on rootstock age; total aggregated height (a), and number of sprouts per stump (b). As age increases, root ASC potential decreases (c).

1.3.5 Impact of cutting height

The location one plot within the MSCNWR was used to examine the effects of stump sprouting with respect to different cutting height treatments. Since DBH was a significant predictor of sprouting response, a multivariate covariance analysis (MACOVA) was conducted. The response variables analyzed include the total aggregated height, the number of sprouts per stump and the stump height ASC. The predictor variables include cutting height (factor) and DBH (continuous). The significance of cutting height decreased with time (Figure 1.15). Letters that are the same above figures 1.16-1.24 indicate no significant differences between the two cutting heights. If the letters are different from one another between cutting heights, there is a significant difference (at *P* = 0.05) between them.

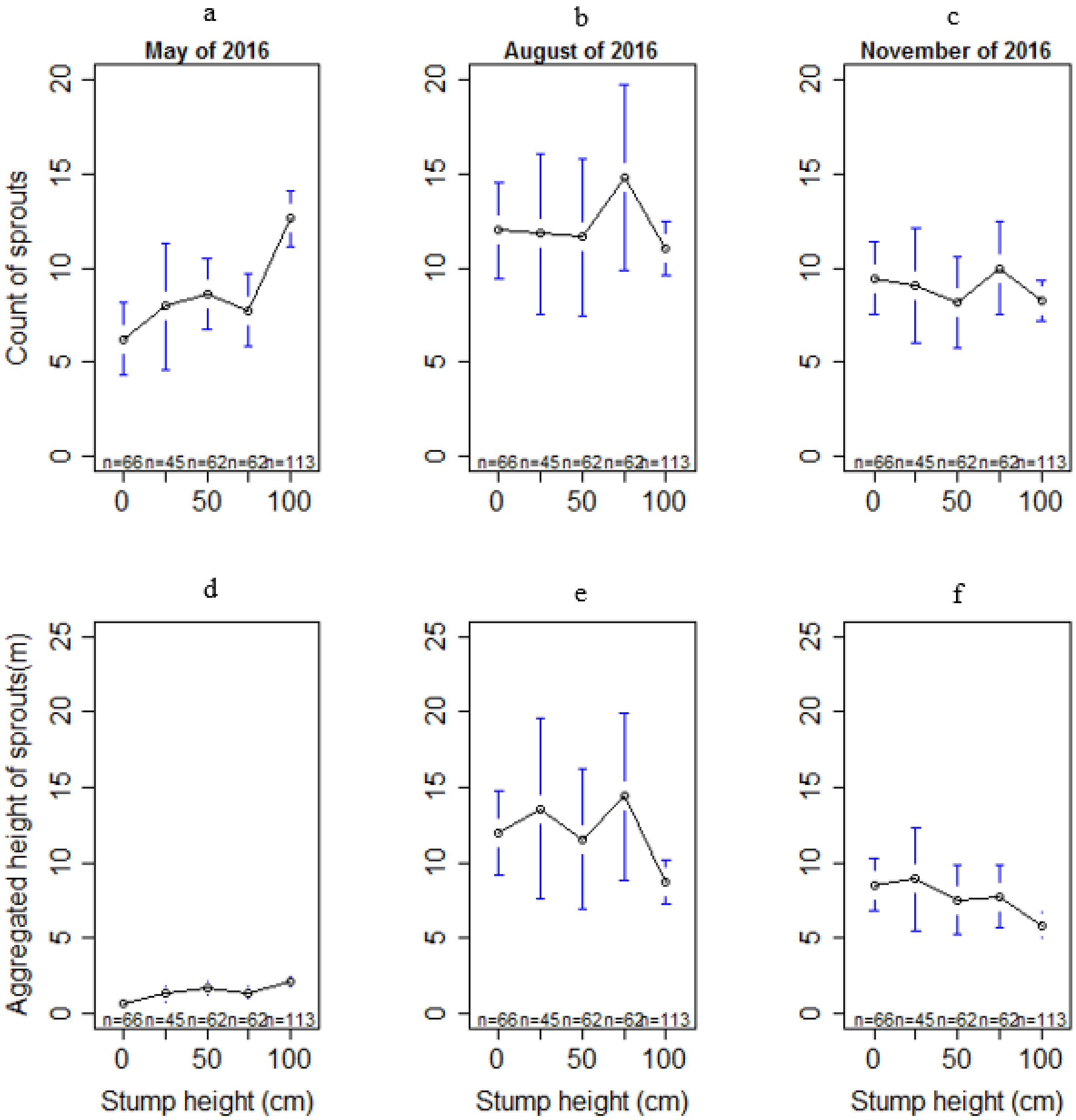


Figure 1.15. The five cutting height levels with average counts of sprouts (a-c) and aggregated height (d-f) per rootstock.

Multiple comparisons on total aggregated height by cutting height

The results of the Pillai test (a test statistic used in a MACOVA; used as suggested by Muller et al., 2007) in the MACOVA indicate cutting height as a significant indicator of aggregated height ($P < 0.001$). Tukey's post-hoc comparisons were used between the five cutting classes and three time intervals. After our first measurement two months post-treatment, 100cm cutting height was significantly greater than 75cm, 25cm, and ground level cutting height in relation to the aggregated total height (Figure 1.16). There was no difference between 100cm and 50cm cutting height ($P = 0.5669$). As time went on, the stump height became less of a significant factor as there were minimal differences between cutting height and aggregated total height five months post treatment (Figure 1.17), while there was no significance eight months post treatment (Figure 1.18).

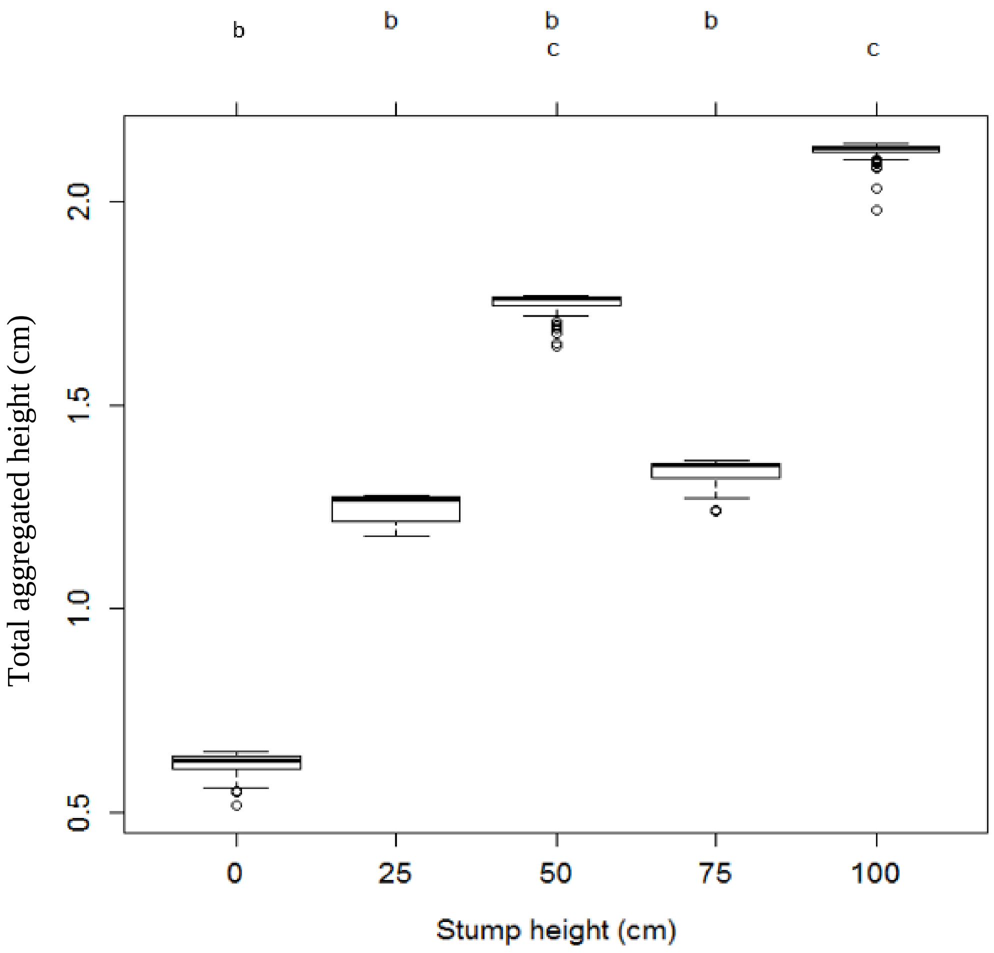


Figure 1.16. Tukey's post-hoc comparisons on total aggregated height by cutting height two months post-treatment. Cutting height 100cm was significantly greater than 75cm, 25cm, and ground level. The differences were noted between ground level and 50cm (*P* = <0.0001), ground level and 100cm (*P* = <0.0001), 100cm and 25cm (*P* = 0.0241), and 75cm to 100cm (*P* = 0.0181).

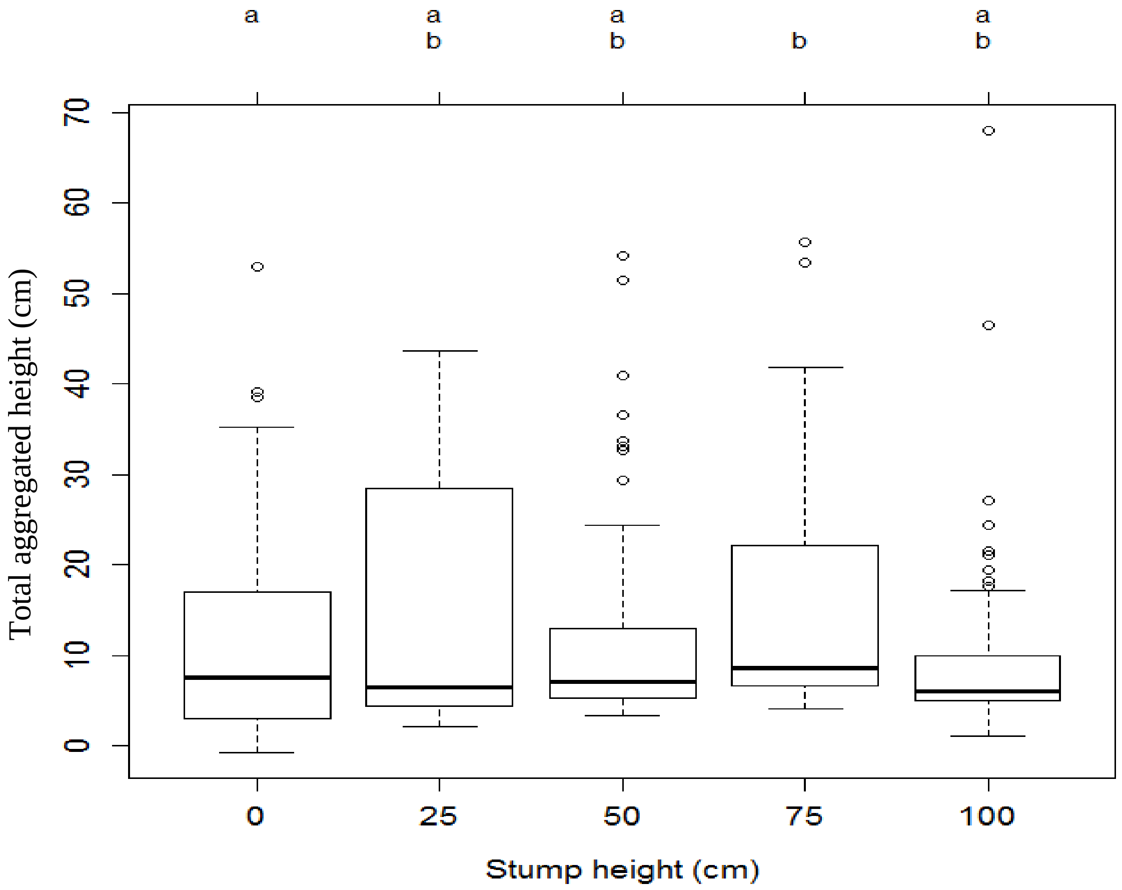


Figure 1.17. Tukey's post-hoc comparisons on total aggregated height by cutting height five months post-treatment. Cutting height became less of an indicator of total aggregated height. The only difference between cutting heights was between ground level and 75cm (*P* = 0.0454).

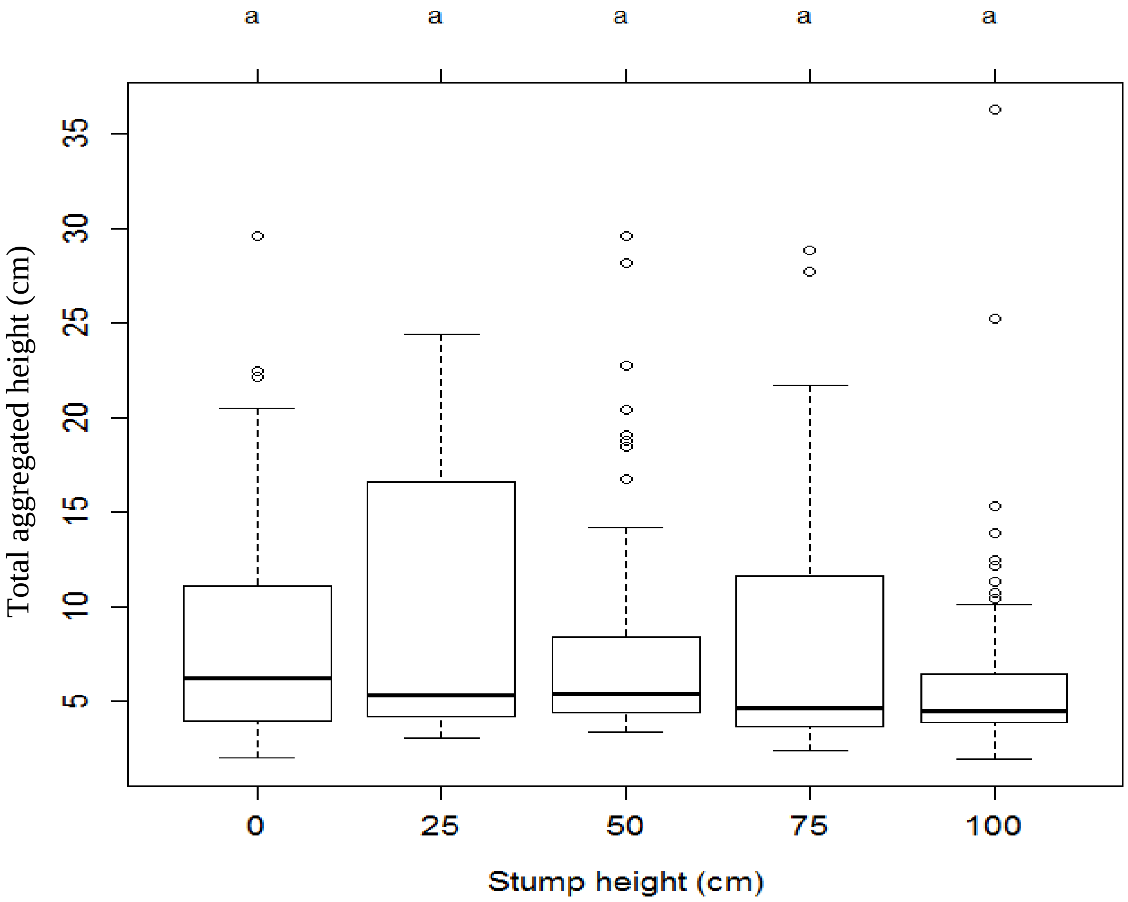


Figure 1.18. Tukey's post-hoc comparisons on total aggregated height by cutting height eight months post-treatment. Eight months post-treatment, cutting height was no indicator of total aggregated sprout height. There was no difference between any classes of cutting height.

Multiple comparisons on total number of sprouts per stump by cutting height

The results of the Pillai test in the MACOVA indicate cutting height as a significant indicator of total number of sprouts per stump ($P < 0.0001$). Tukey's post-hoc comparisons were used between the five cutting classes and three time intervals. Two months post-treatment, 100cm cutting height class was significantly greater than all other classes except for 50cm ($P = 0.05123$; Figure 1.19). Five months post-treatment, the effects of cutting height were smaller than two-months, where the only differences were between ground level and 75cm (*P* = 0.0176) and 75cm and 25cm (*P* = 0.0396; Figure 1.20). Eight months post-treatment, the differences between ground level and 75cm and 75cm and 25cm remained as they did in the five-month post-treatment measurement (Figure 1.21).

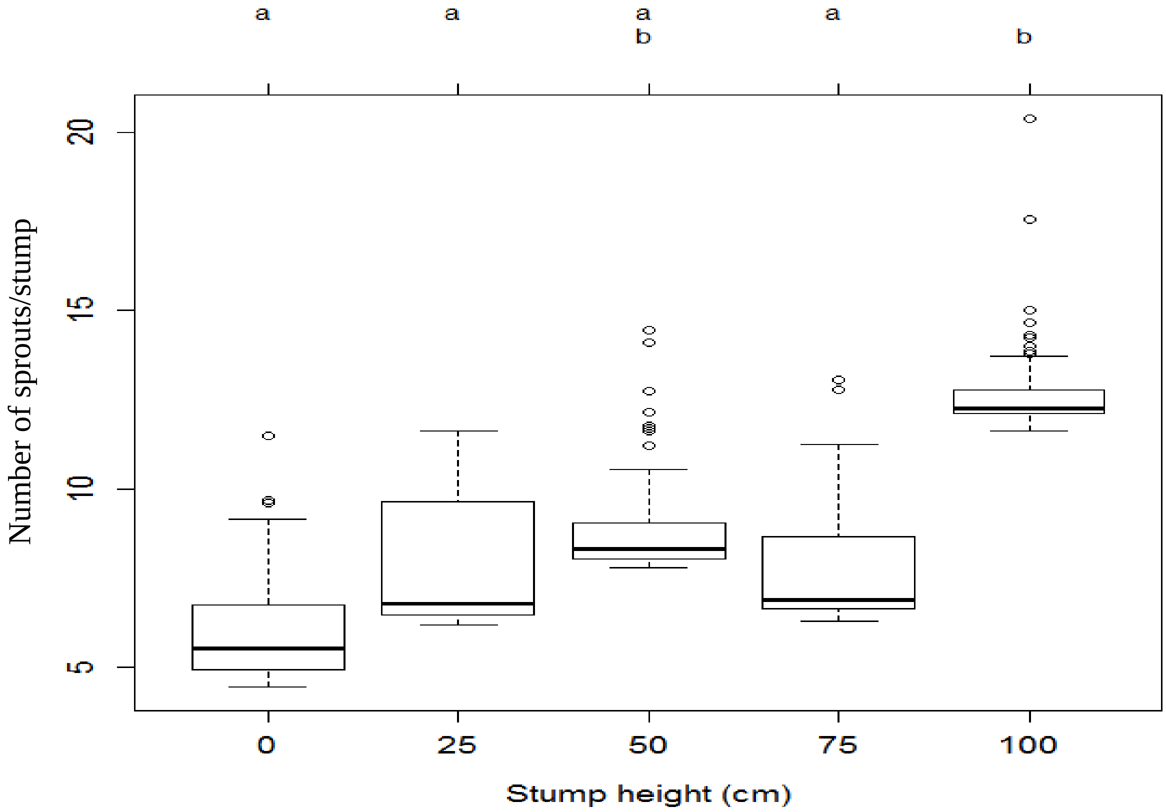


Figure 1.19. Tukey's post-hoc comparisons on total number of sprouts per stump by cutting height two months post-treatment. There was significance in 100cm stump height having a greater total number of sprouts per stump versus 75cm (*P* = 0.00194), 25cm (*P* = 0.00467), and ground level heights (*P* = < 0.001). There was no difference between 100cm and 50cm heights (*P* = 0.05123).

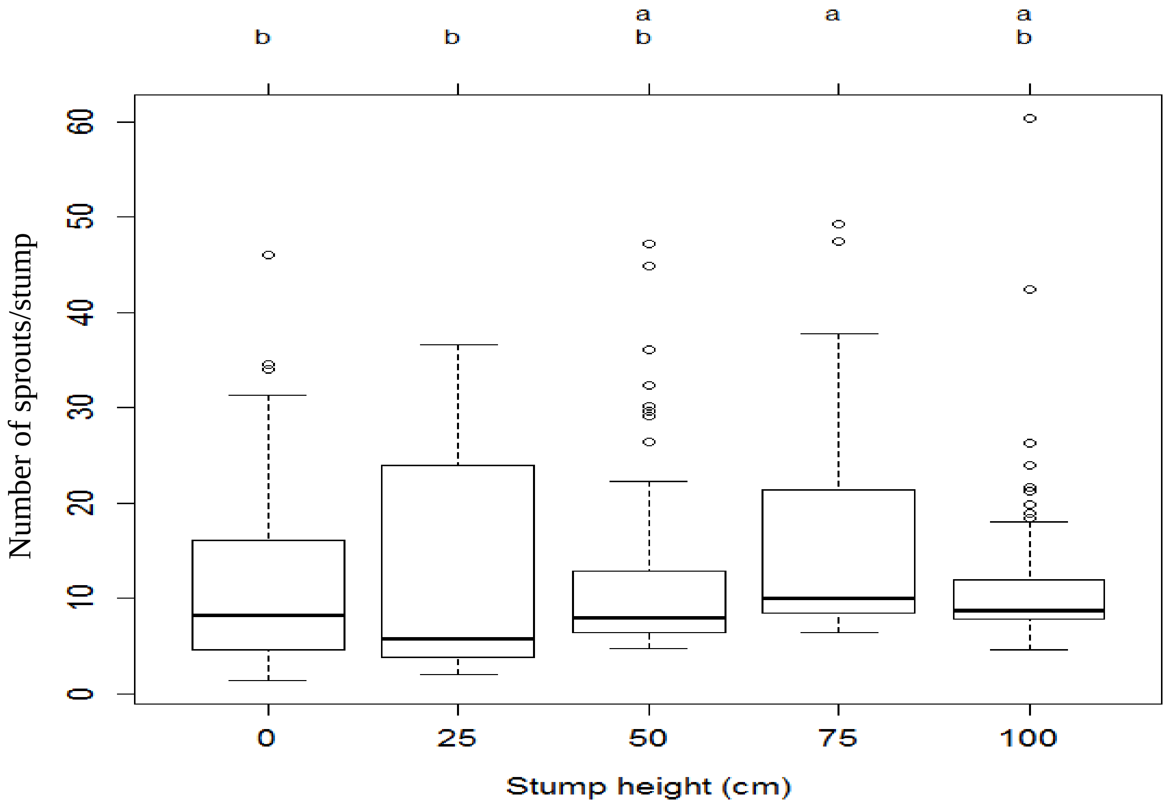


Figure 1.20. Tukey's post-hoc comparisons on total number of sprouts per stump by cutting height five month post-treatment. The only difference was between ground level and 75cm (*P* = 0.0176), and 75cm and 25cm (*P* = 0.0396).

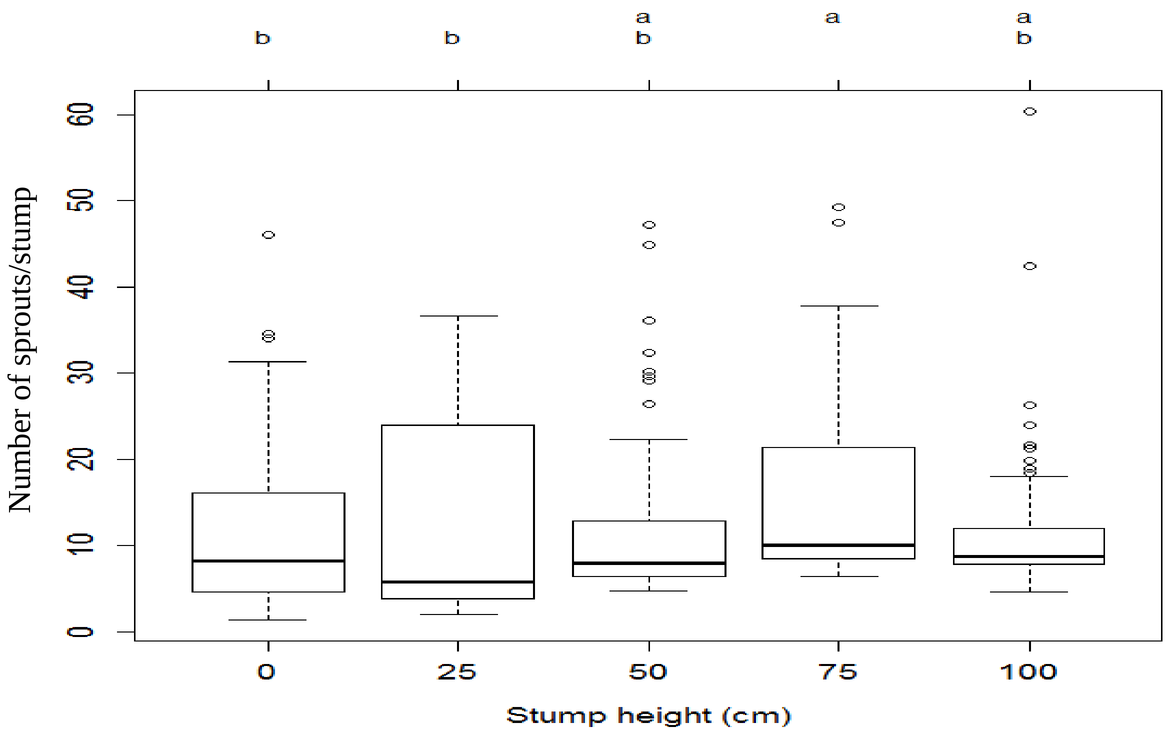


Figure 1.21. Tukey's post-hoc comparisons on total number of sprouts per stump by cutting height eight months post-treatment. The only difference was between ground level and 75cm (*P* = 0.0176), and 75cm and 25cm (*P* = 0.0396).

Multiple comparisons on stump height ASC by cutting height

The last comparisons regarding cutting height involved the transformed stump height ASC. Tukey's post-hoc comparisons were used between the five cutting classes and three time intervals. Two months post-treatment, the only significant differences were between ground level and 75cm (*P* = 0.0335), and ground level and 100cm (*P* = 0.0215; Figure 1.22). Five months post-treatment, 100cm, 75cm, 50cm, and 25cm were all significantly lower (*P* = <0.001 for all four) than ground level cutting height (Figure 1.23). Eight months post-treatment, cutting height 0 was greater than all four cutting heights, and cutting height of 100cm was significantly lower than both ground level ($P < 0.001$) and 25cm ($P = 0.00213$; Figure 1.24). 25cm, 50cm, and 75cm did not differ from one another.

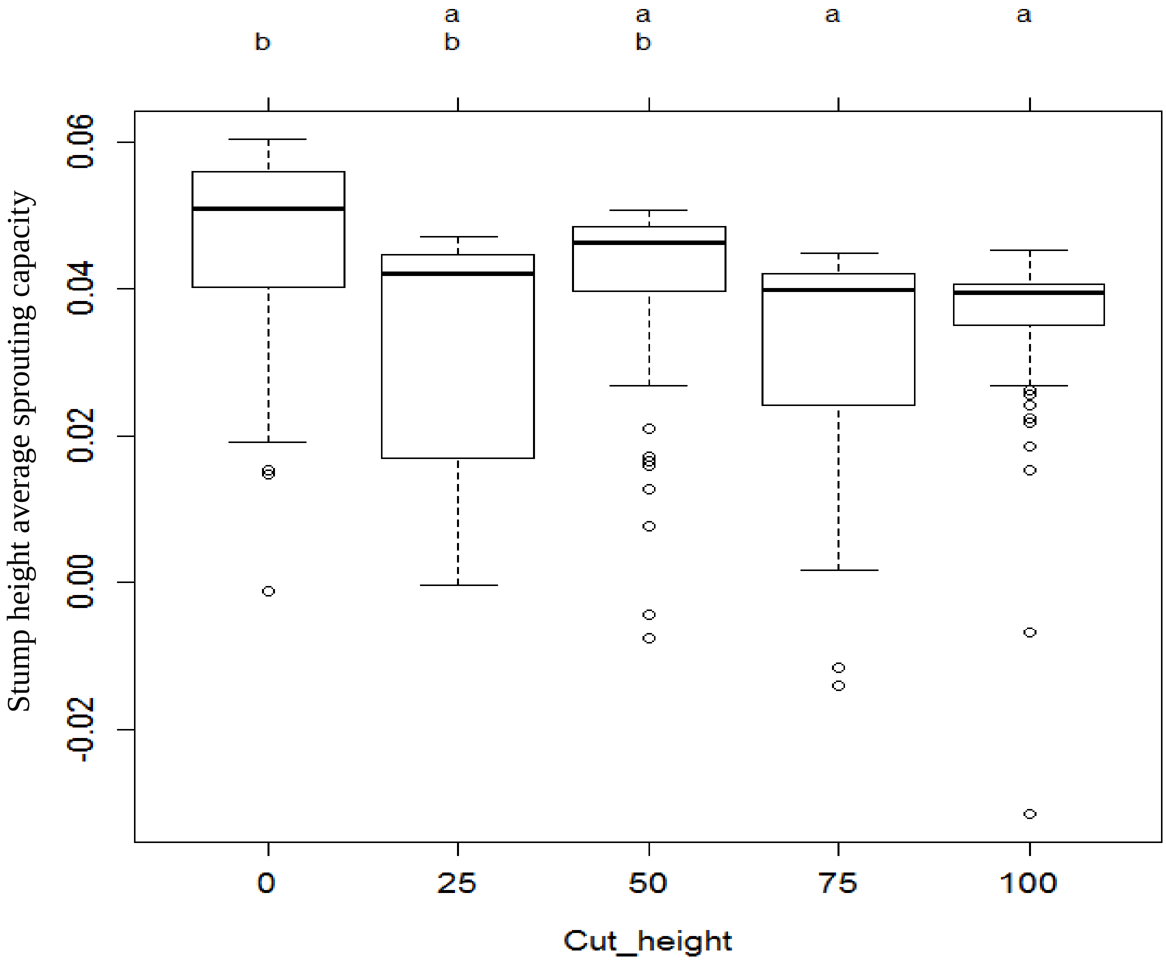


Figure 1.22. Tukey’s post-hoc comparisons on stump height ASC by cutting height two months post-treatment. Ground level was significantly higher than 75cm and 100cm. All others (25cm, 50cm, 75cm, 100cm) were equal to one another ($P > 0.05$ for all).

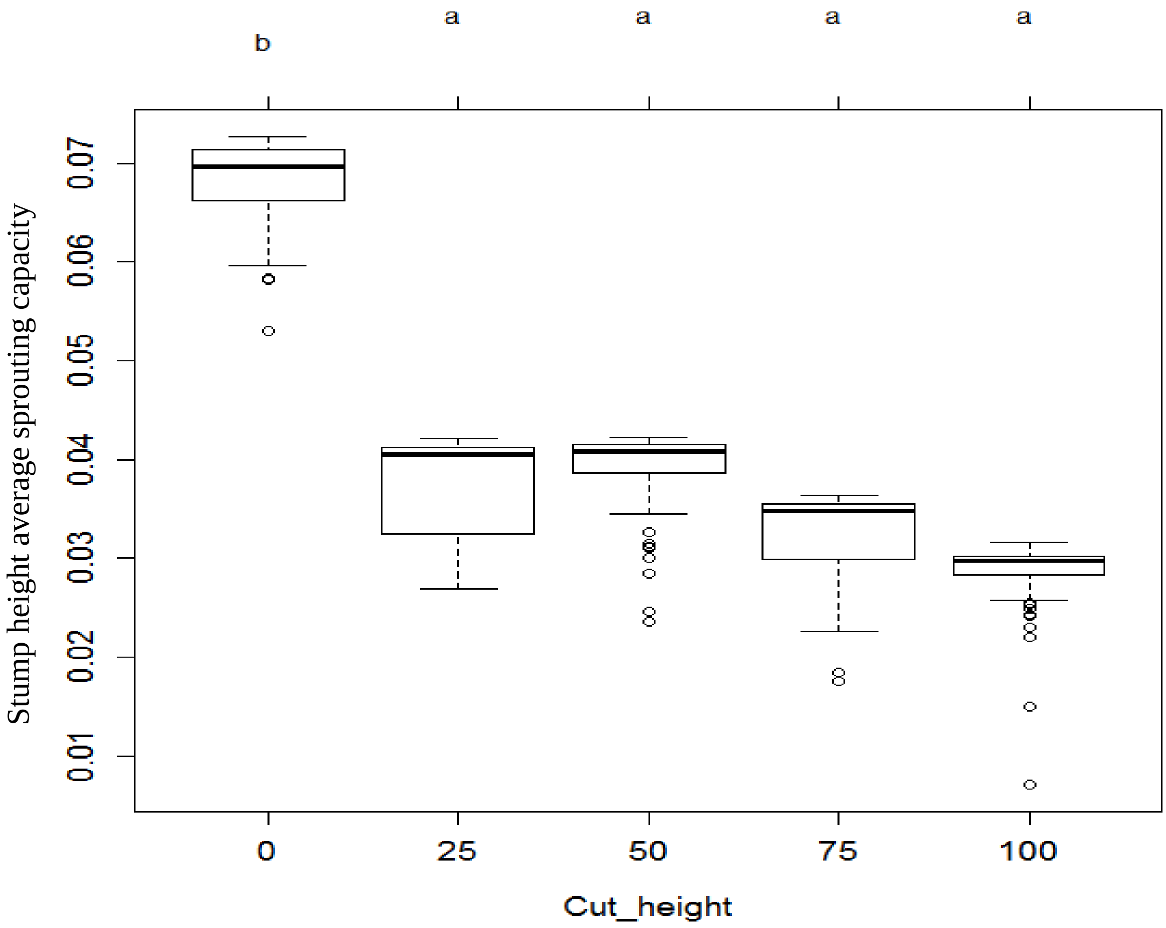


Figure 1.23. Tukey's post-hoc comparisons on stump height ASC by cutting height five months post-treatment. Ground level was significantly higher than the four other cutting heights. All others were equal to one another ($P > 0.05$).

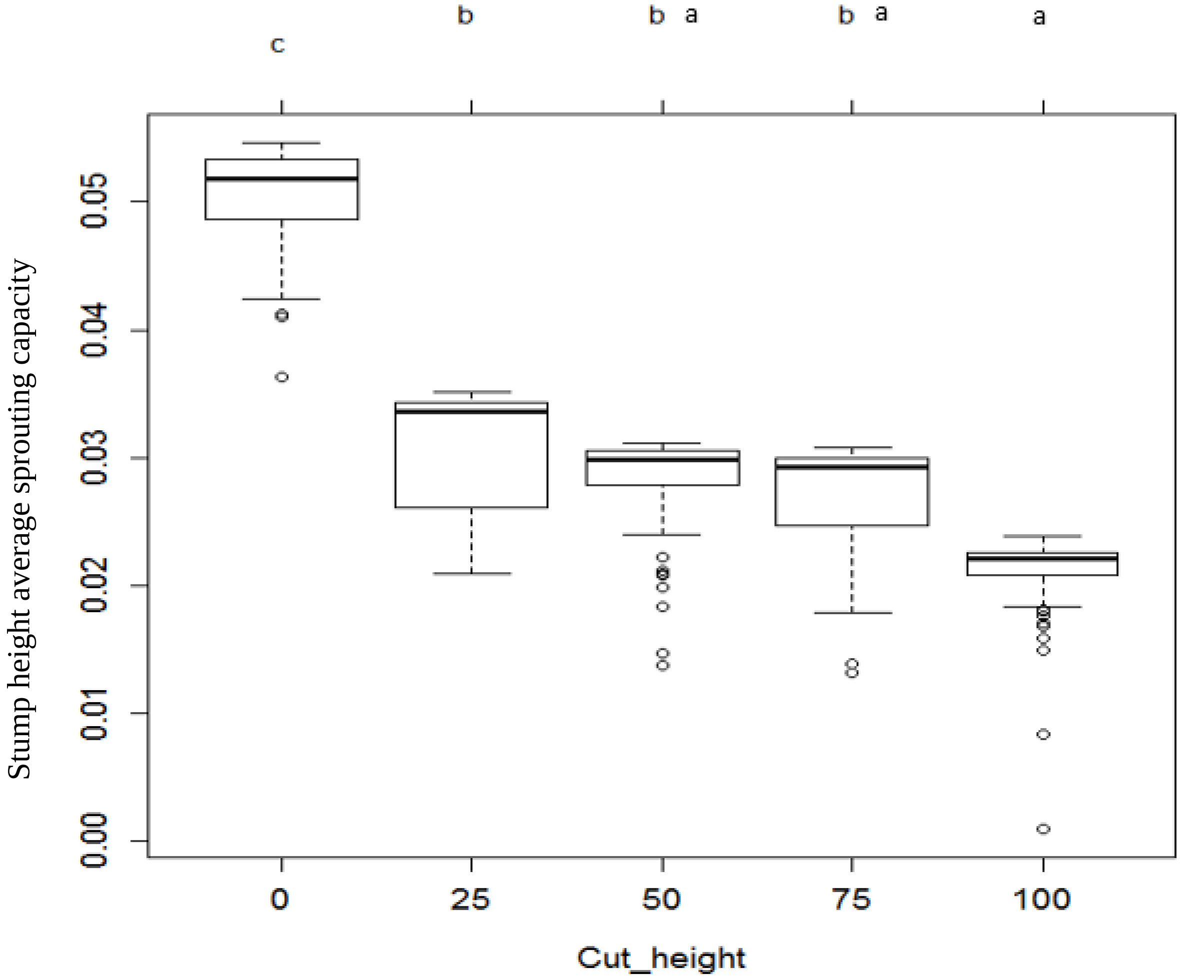


Figure 1.24. Tukey's post-hoc comparisons on stump height ASC by cutting height eight months post-treatment. Ground level was significantly higher than all other cutting height classes.

1.3.6 Impact of prescribed fire

Location one at the MSCNWR (stump heights <75cm), as well as low and medium canopy closure from the GBNERR, in addition to all data from location two within the MSCNWR were used for analysis while evaluating the impact prescribed fire had on total aggregated height and the total number of sprouts per stump. Since stump diameter was significant in determining the total number of sprouts and total aggregated height, stumps were broken into three size categories; 0-5cm DBH, 5-10cm DBH, and >10cm DBH. Two sample t-tests were ran for both total number of sprouts and total aggregated height for each DBH class. Results are summarized in Table 1.6 below.

Table 1.6. The effects of prescribed fire on total aggregated height per stump (cm) and the total number of sprouts on each stump were compared using two sample t-tests. There were three DBH classes.

DBH class	Test	P-value	Comparison				
0-5cm	Aggregated height	0.981		N	Mean	StDev	SE Mean
			No fire	80	164	117	13
			Fire	64	164	109	14
0-5cm	Total # sprouts	0.704		N	Mean	StDev	SE Mean
			No fire	80	5.58	3.56	0.40
			Fire	64	5.30	4.89	0.61
5-10cm	Aggregated height	0.004		N	Mean	StDev	SE Mean
			No fire	47	267	231	34
			Fire	45	396	184	27
5-10cm	Total # sprouts	<0.001		N	Mean	StDev	SE Mean
			No fire	47	8.47	6.80	0.99
			Fire	45	15.29	8.16	1.20
>10cm	Aggregated height	0.598		N	Mean	StDev	SE Mean
			No fire	44	703	535	81
			Fire	47	654	311	45
>10cm	Total # sprouts	0.248		N	Mean	StDev	SE Mean
			No fire	44	20.5	17.6	2.7
			Fire	47	24.3	13.7	2.0

1.4 Discussion

1.4.1 Sprouting capability

Chinese tallow tree has strong sprouting capabilities and multiple factors are likely to affect that. Prescribed fire, canopy closure, stump diameter, tree age, and stump cutting height all influence the response of stump sprouting after felling treatments.

Many stumps began sprouting vigorously just a few weeks after cutting. Self-thinning was observed between the five- and eight-month post-treatment measurement, indicated by the declining total number of sprouts per stump (Figure 1.15; a,b,c). Our data indicate the number of sprouts per stump declining by nearly half between the five- and eight-month post-treatment measurements. Scheld and Cowles (1981) reported similar self-thinning trends in Chinese tallow with stumps initially having 12-24 sprouts each, with self-thinning reducing that number to 4-8 at the end of the second growing season. Self-thinning of stump sprouts is a common process observed in many different tree genus such as *Quercus* and *Sequoia* (Mitchell et al., 1985; O'Hara and Berrill, 2010).

1.4.2 Impact of prescribed fire

Prescribed fire showed no impact on our trees greater than 10cm DBH. This is likely because of increased bark thickness as the tree gets older/larger (Meyer, 2011). Trees less than 10cm in DBH but greater than 5cm DBH exhibited increased growth and increase in total number of sprouts when exposed to fire compared to those not exposed. Trees less than 5cm DBH showed similar results as the large trees, where fire did not affect either the total number of sprouts per stump or total aggregated height of each stump. Prescribed fire has shown to have minimal effect of controlling established Chinese tallow (Meyer, 2011). Larger trees survive fire, whereas smaller ones are top-killed and exhibit vigorous sprouting capabilities (Grace et al., 2005; Meyer, 2011). Our stumps affected by fire did not exhibit any sprouting from the trunk or top of the stump, but instead at the base and along the root line. Our observations in the field align with those of Grace (1998) that show roots extremely capable of sprouting, sometimes up to five meters away from the original tree stump. Our data one growing season post-burn show no effects of fire on stump survival. In fact, there was a 98% survival rate where a stump was affected by fire. Similar results have been reported by Grace (1998) where the only mortality occurred from transplanted seedlings into the burn site. It was reported that no mortality occurred from the naturalized population, indicating a 100% survival rate (Grace, 1998). Two growing seasons post-fire, Grace's (1998) preliminary data show a significant decline in tree survival; where all new seedlings, 70% of trees 2 meters or less in height, and 30 trees ranging from 2-5 meters were either killed or top-killed. Based off our data, we cannot suggest fire should be used as an idealized control mechanism of already invaded areas. However, fire was shown to be effective at minimizing the risk of invasion at a site in northern Florida (Burns and Miller, 2004).

1.4.3 Impact of canopy closure

Canopy closure was significant in determining the total number of sprouts per stump and the average root sprouting capacity. High canopy closure areas led to a lesser amount of sprouts per stump than the intermediate and low canopy closure areas. This is consistent with non-disturbed Chinese tallow trees, where tallow survives and grows better in open sunlit areas (Meyer, 2011). Additionally, our results show a decrease in average root sprouting capacity with increasing canopy closure. Our high canopy closure site experienced a mortality rate of nearly 22%, compared to 1.7% and 1.1% for the intermediate and low canopy closure sites, respectively. Lin et al., (2004) reported a mortality rate of 5.3% when the trees were exposed to only 0.1% full sunlight. That is in contrast to their 10% full sunlight area where mortality was experienced at a rate of 0.05%. Another study (Siemann and Rogers, 2003) showed significantly lower Chinese tallow seedling survival in shade treatments compared to higher sunlight treatments. In a study comparing seedling survival, Bruce (1993) observed a 100% mortality rate when Chinese tallow seedlings were in a full-shade environment. Based off our data, forests/prairies with an open canopy are at major risk of not only invasion, but also dominance by Chinese tallow. Stump sprouting was not controlled in areas with a low canopy percentage. Forests with a dense canopy are still at risk of invasion by Chinese tallow, however; the impacts will not be as significant. Some stumps will re-sprout where others will die, and the ones that do re-sprout will be small enough in height that minor additional mechanical treatments will possibly eradicate the re-sprout.

1.4.4 Impact of tree age and stump diameter

Stump diameter (GLD) and DBH are common measurements used to report the initial size of a stump/tree in many studies (Ducrey and Turrel, 1992; Randall et al., 2005; Ashish et al., 2010). The larger a stump, whether based off GLD or DBH usually indicate an older tree (Blake, 1981), where other studies (MacDonald and Powell, 1983; Khan and Tripathi, 1986; Ashish et al., 2010) have explained the effect that stump size has on sprouting effects due to the changing physiology of older trees. Our data show similar trends that as the larger the GLD or DBH is, the older the tree is. Therefore, the same conclusions can be drawn between tree age and/or stump diameter.

Our data show that as stump diameter increases in size, the total number of sprouts per stump and the total aggregated height increases. Stump diameter was potentially the most significant indicator of stump sprouting in our study (indicated by $P = <0.001$). Bigger stumps could support more sprouts, as well as more total biomass (total aggregated height) than smaller stumps. Scheld and Cowles (1981) reported similar trends; their largest stumps produced the greatest total number of sprouts, as well as the tallest height of the dominant sprout. This pattern is reported in certain *Quercus* species as well (Johnson, 1975; Wu et al., 2008). Scheld and Cowles (1981) reported their largest stumps having a dominant sprout of 12 feet within one growing season, and more than 18 feet after the second growing season. This phenomenon may be attributed to the fact that larger stumps have more surface area, a more advanced root system, and a large amount of stored carbohydrates that allow larger stumps to provide more energy and space for sprouts (Miura and Yamamoto, 2003; Yamada and Suzuki, 2004). Controlling the larger stumps from sprouting should be of high importance to land managers as they have the highest potential of detrimental effects on any native or important vegetation.

1.4.5 Impact of cutting height

Stump height initially had an effect on the total number of sprouts per stump (Figure 1.19) with the highest cutting height being able to support the most number of sprouts per stump, however, that trend disappeared five and eight months after initial treatment (Figures 1.20; 1.21). A negative relationship existed with cutting height and average sprouting capacity of a stump (Stump Height ASC; Figures 1.22-1.24). No studies examining the effect of cutting height on Chinese tallow stump sprouting exists to our knowledge; however, similar trends were reported in *Quercus variabilis*, with Xue et al. (2013) reporting cutting height as a significant factor on sprout number and sprout growth during the first year after cutting and diminishing effects after that.

The initial significance of cutting height and total number of sprouts per stump could be attributed to the sprouts originating from dormant adventitious buds. The higher cutting height meant more surface area aboveground, therefore; a greater number of dormant buds were accessible to sprout (Harrington, 1984). After the initial advantage due to more adventitious buds and a larger amount of aboveground biomass, the sprouts may have come more dependent on the developed root systems where there is much more nutrient availability, which could have been the reason cutting height was only a significant factor on total number of sprouts for the first few months before that significance disappeared (Xue et al., 2013).

1.5 Conclusions

Our results show that Chinese tallow exhibits extremely vigorous stump-sprouting capabilities. The amount of stump-sprouting is largely dependent on the size of the tree and its position with respect to canopy characteristics. Our high canopy closure sites exhibited the most mortality, nearly 40% higher than the low canopy closure areas. Applying a mechanical treatment on trees in high canopy closure areas may minimize the sprouting ability of this invasive. However, areas in high sunlight/open areas have a high possibility of invasion and minimal control of stump-sprouting without any herbicidal treatment. Fire had no effect on trees larger than 10cm DBH, but worked as a growth enhancer in trees smaller than 10cm. The small trees affected by fire responded and had larger stems than equal sized ones in non-burned areas. Cutting height had an initial positive relationship with the total number of sprouts, but that effect disappeared five to eight months post-treatment. Complete control without herbicides is unlikely.

In weed science when evaluating herbicide control, anything less than 85% mortality is considered a failure. We were nowhere near that; therefore, we do not recommend mechanical treatment as a feasible method of control of Chinese tallow.

We will continue to monitor stumps affected by fire prior to felling to see the long-term effects fire has on stump survival/sprouting. Future research should examine the effects of prescribed fire on Chinese tallow stump sprouting in high canopy closure areas. This could be an area where herbicide treatment may not be necessary to control the impact of this invasive.

Chapter 2

Evaluating longleaf pine (*Pinus palustris*) natural regeneration 15-years post regeneration control

2.1 Introduction

Arguably the most important tree species dependent ecosystem in the southern United States, the longleaf pine (*Pinus palustris* Mill.) ecosystem has experienced a drastic reduction in area, from approximately 37.2 million hectares (92 million acres) pre-European settlement (Frost, 2006), to today's 1.3 million hectares (3.3 million acres) (Oswalt et al., 2012). Due to its ground layer being dominated by grasses and a diversity of forbs (Oswalt et al., 2012), longleaf ecosystems provide habitat for fauna that many other ecosystems cannot (Oswalt et al., 2012). Characterized as a park-like, intermediate in density savanna (McGuire et al., 2001; Perkins et al., 2008) longleaf ecosystems support hundreds of plants and animal species such as the red-cockaded woodpecker (Alavalapati et al., 2002). 1 m^2 can contain more than 40 different vascular plant species (Walker and Peet, 1983). This is, in part, because this ecosystem evolved with frequent, natural ground fires (Croker and Boyer, 1975). Longleaf pine forests are home to 389 rare plant taxa, out of which 187 are considered rare and/or vulnerable to extinction (Walker, 1993). The decline of the longleaf ecosystem can be attributed to several key factors; a century-long period of logging, land conversion (e.g., urbanization, tree plantations, agriculture), regeneration failures, and fire suppression following European settlement significantly reshaped the historical vegetation/fuel structures and fire regimes (Croker and Boyer, 1975; Van Lear et al., 2005). Since fire suppression is recognized as a large cause of decline of longleaf pine habitat, it should be no wonder that many of the nearly 400 rare plant species depend on fire to survive (Glitzenstein et al., 2001).

The use of prescribed fire in combination with vegetation management practices, (e.g., removal of overstory and midstory trees) has been recognized as a preferred approach to restoring historical longleaf pine ecosystems and wet pine savannas, reducing hazardous fuels and subduing large, destructive wildfires that threaten interests in the southeastern region (Lavoie et al., 2010). Without the use of prescribed fire in the longleaf ecosystem, litter and woody components increase, species richness declines, and the understory becomes dense and closes (Palmquist et al., 2015). A growing interest in restoration of this species is occurring (Brockway and Outcalt, 2000; Jose et al., 2006), and although natural regeneration of this species is desirable in practice, it has been proven difficult to achieve in the field (Croker and Boyer, 1975).

Croker and Boyer (1975) characterize regeneration failures as a leading cause of the decline of the longleaf pine coverage. Regeneration failures can be attributed to a long gap between seed years and/or competition. The density of the parent plants (Harper, 1977; Grace and Platt, 1995) affects recruitment of new plants. This is because of competition between and within species for sunlight, nutrients, and other resources (Curtis, 1959; Spurr and Barnes, 1980; Oliver and Larson, 1990; Grace and Platt, 1995). Platt et al. (1988) and Platt & Rathbun (1994) found that recruitment of new longleaf pine seedlings was minimal in areas that had high adult tree densities.

Natural regeneration of longleaf pine can be achieved, but must be done with careful precision, and is most effective with the shelterwood silvicultural system (Croker and Boyer, 1975). It has been recommended that longleaf ecosystems be thinned to a basal area of 30 ft^2/acre when an area is ready to be regenerated (Croker, 1973). Basal area management is often times the basis for decision making regarding the regeneration needs of a forest (Elledge and Barlow, 2012). Defined as the cross-sectional area of a tree (usually in square feet) measured at breast height (4.5 feet tall), basal area is an important management tool used by land managers and silviculturists alike (Elledge and Barlow, 2012).

Some studies show a zone of exclusion under mature longleaf pine trees where there is a significant reduction in the amount of seedlings present (Wahlenberg, 1946; Smith, 1955; Brockway and Outcalt, 1998). There have been two hypotheses to explain this phenomenon; the first being higher levels of sunlight availability near the center of gaps, and the second being direct needle fall beneath the large mature trees leading to more intense prescribed fires and killing of the seedlings (Boyer, 1974; Platt et al., 1988; Grace and Platt, 1995; Farrar, 1996). However, Brockway and Outcalt (1998) show evidence that neither of those hypotheses are a plausible explanation for this zone of exclusion. Brockway and Outcalt (1998) concluded that the most likely mechanism causing the zone of exclusion was due to the competition between the root systems of seedlings and adults. Contrasting evidence shows that individual trees may increase the amount of soil nutrients and soil moisture available to the understory in a savanna-type environment (McGuire et al., 2001). These two factors should limit any intraspecific competition between the seedlings and adult trees.

Due to the limited zone of exclusion we observed in the field, we were motivated to question whether the zone of exclusion previously mentioned exists as frequently as reported. McGuire et al. (2001) suggests longleaf pine overstory increases, rather than decreases, the survival and establishment of longleaf pine seedlings. It also has been reported that naturally regenerated seedlings can be frequently found within 4 meters of an adult mature longleaf pine

(McGuire et al., 2001). Walker and Davis (1956) showed that there were only 13% less seedlings greater than 4.8 meters from a mature tree compared to a canopy gap. In part of these two conflicting conclusions of whether there is a clear zone of exclusion, more research should be done to better understand this phenomenon. In fact, the relationship between adult longleaf pine trees and their regeneration is not well understood (McGuire et al., 2001).

Our study investigated the regeneration of longleaf pine in a managed area in southern Alabama. Having our study site at an area that has been managed for longleaf pine since the 1940's gave us a unique opportunity to assess what situations prohibit/enhance regeneration of this species. The first hypothesis we tested was that there was no difference in amount of regeneration in regards to different basal area classifications. The second hypothesis we tested was there were no small trees (regeneration) under large, mature trees. Specifically, we hypothesize that 1) there will be a significant difference in the amount of regeneration from the different basal area classes, and 2) there will be regeneration under large, mature trees, despite previous research showing a 16 meter "zone of exclusion" under/around mature trees. These results will help land managers identify more appropriate management techniques of this critical species.

2.2 Materials and methods

2.2.1 Regional longleaf pine growth study design

In 1964, the U.S. Forest Service established the Regional Longleaf Pine Growth Study (RLGS) in the Gulf States (Farrar, 1978). The original objective of the study was to obtain a database for the development of growth and yield predictions for naturally regenerated, even-aged longleaf pine stands. Plots were installed to cover a range of ages, densities, and site qualities. The RLGS accounts for possible growth change over time by adding a new set of plots in the youngest age class every 10 years. The project completed its ninth re-measurement period (45-year measurement) during spring 2007.

The study consists of 305 permanent 0.04 and 0.08 ha (1/10- and 1/5-acre) measurement plots located in central and southern Alabama, southern Mississippi, southwest Georgia, northern Florida, and the sandhills of North Carolina. There are five stand age classes ranging from 20 to 120 years, five site-index classes ranging from 15.2 to 27.4 m (50 to 90 feet) at 50 years, and six density classes ranging from 7 to 34 m^2/ha (30 to 150 ft^2/acre), plus plots left free to grow. Densities are established and maintained by thinning from below, where the most suppressed trees are taken out. Within this distribution are five time replications of the youngest age class. All five replications are located on the Escambia Experimental Forest (EEF) in Brewton, AL.

Net (measurement) plots are circular and 0.08 ha (1/5-acre) in size surrounded by a similar and like-treated 1/2-chain (10 m) wide isolation strip with both surrounded by a 1/2-chain wide protective buffer strip that receives extensive management. Plots are inventoried, and treated as needed, every 5 years. The measurements are made during the dormant season (October through March) and it takes three years to complete a full re-measurement of all plots.

Data recorded for every net plot tree with a diameter breast height (DBH) > 1.27 cm include: tree number, tree azimuth, tree distance, DBH, crown class, and wood pole information (i.e., height to 6-inch top diameter). A systematic sub-sample of trees from each one-inch dbh class has been permanently selected and measured for height to the live-crown base, total height, and age.

2.2.2 Current study design and study site

This study took place at the Escambia Experimental Forest, in southern Alabama (31°00’N – 87°03’W). This site is located near the center of the native range of longleaf pine distribution. The EEF is a 3000 acre tract of land which was established in 1947 as a cooperative effort between the United States Forest Service in conjunction with T.R. Miller Mill Company to study longleaf pine forests as a whole, with a focus on natural regeneration (Barlow et al., 2011).

Primary management of the EEF is through the use of prescribed fire, with every unit having a fire return interval of no greater than three years. The basal area of each plot is maintained in accordance with its initial target by periodic thinning from below methods. The majority of the land is dominated by longleaf pine forests, with just a few hardwood areas near water (i.e., stream, pond)

Since the original objective of these plots was to develop growth and yield models, all regeneration in the plots were mowed over up until 2001, giving us an ideal opportunity to explore the regeneration patterns of the longleaf pine. Data collection occurred during June, 2016. The study consisted of measuring 40 1/5th acre RLGS plots. The number of each plot density that we measured is summarized in Table 2.1.

Table 2.1. Plot density of longleaf pine and number measured for each density.

Plot basal area (ft^2 / acre)	Number of each plot
30	11
60	10
90	10
120	7
150	2

In order to examine the location, distance, and amount of regeneration in proximity to the overstory trees, we mapped, and measured, all longleaf that were in the grass stage and beyond. For the nature of this study, a seedling was defined as a first year seedling that germinated this year from seed. Additionally, a sapling was defined as anything in a grass stage or greater (that was not previously marked as a mature tree). Data collected included total height, DBH (if applicable), azimuth, and distance from plot center. In addition, 1/1000th acre strips were used to look at seedling density (Figure 2.1). Those were conducted on a north, east, south, and west pattern, always measuring on the clockwise side of the tape. The 1/1000th acre strip began six feet from plot center, allowing us to not walk over the seedlings in the strip. The strip was 29' long and 18" wide. Data collected included distance on X and Y axis to allow us to find those seedlings again for future survival measurements.

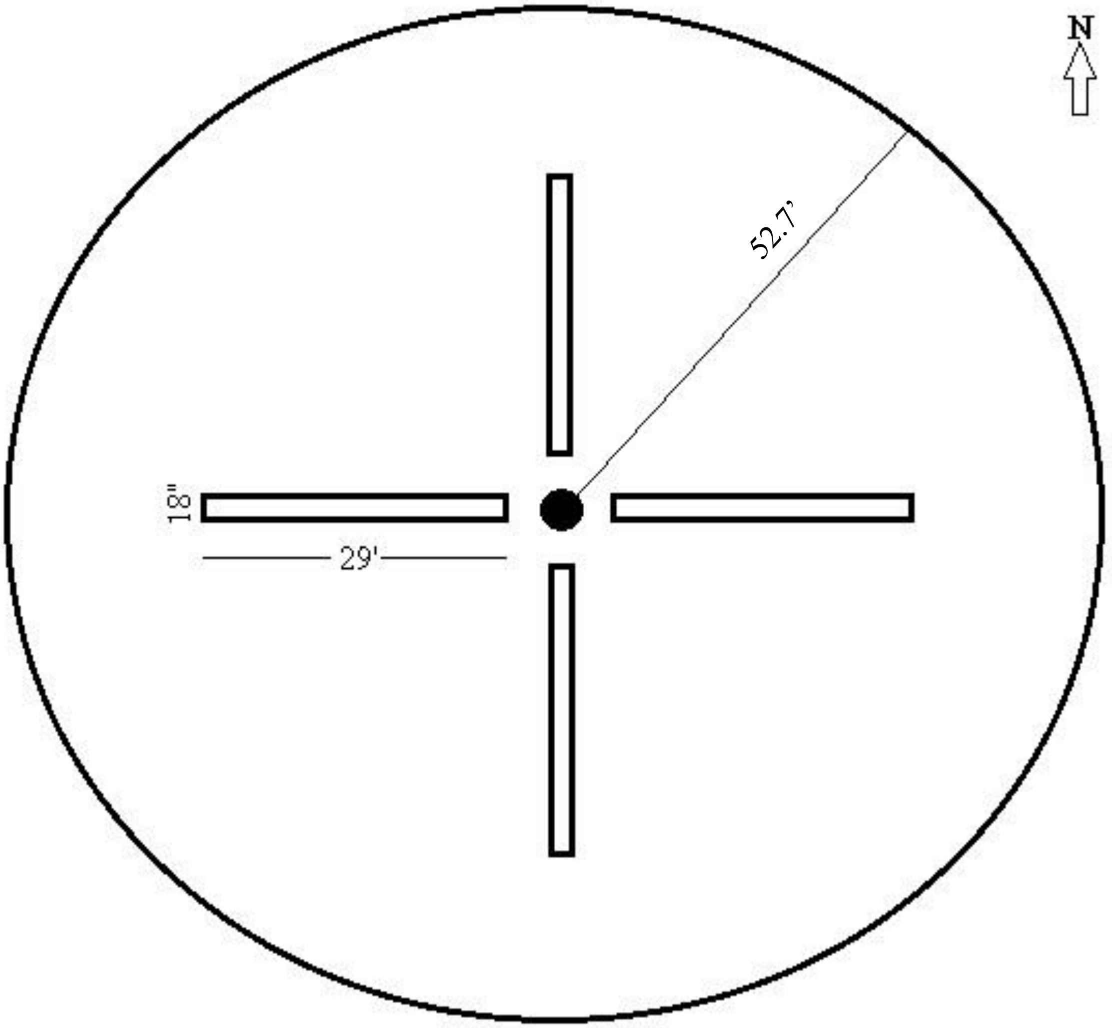


Figure 2.1. 0.08 ha (1/5th acre) plot was used to examine the longleaf pine tree sapling regeneration. In addition, four (N, E, S, W) 1/1000th acre plots were used to examine the seedling density and location.

2.2.3 Data analyses

Seedling and sapling-level analysis

The following variables were used to analyze the hypothesis regarding the amount of regeneration (seedlings/acre) and number of saplings/acre based on different basal area classes: stand basal area, stand basal area management class, quadratic mean diameter (QMD), overstory tree age, tree density (trees/acre), site index, and time since last fire. The response variables used included seedlings/acre and saplings/acre. Since our data were highly skewed and had a large amount of ‘zeros’ for our response variable (seedlings/acre, saplings/acre; Figures 2.2-2.3), a zero inflated negative binomial regression was conducted. Running this test did two things. The first result was the count model which told us what each continuous predictor variable (i.e., time since last fire, basal area, QMD, overstory tree age) was doing to the response variable (i.e., seedlings/acre, saplings/acre). In other words, it allowed us to predict the number of seedlings and/or saplings/acre as a function of different stand conditions. The second result generated was the binary model which predicted the probability of the response variable (seedlings/acre, saplings/acre) being a zero. Comparing Akaike information criterion (AIC) between this model and other models indicated this model being the best option for analysis based off the lower AIC values.

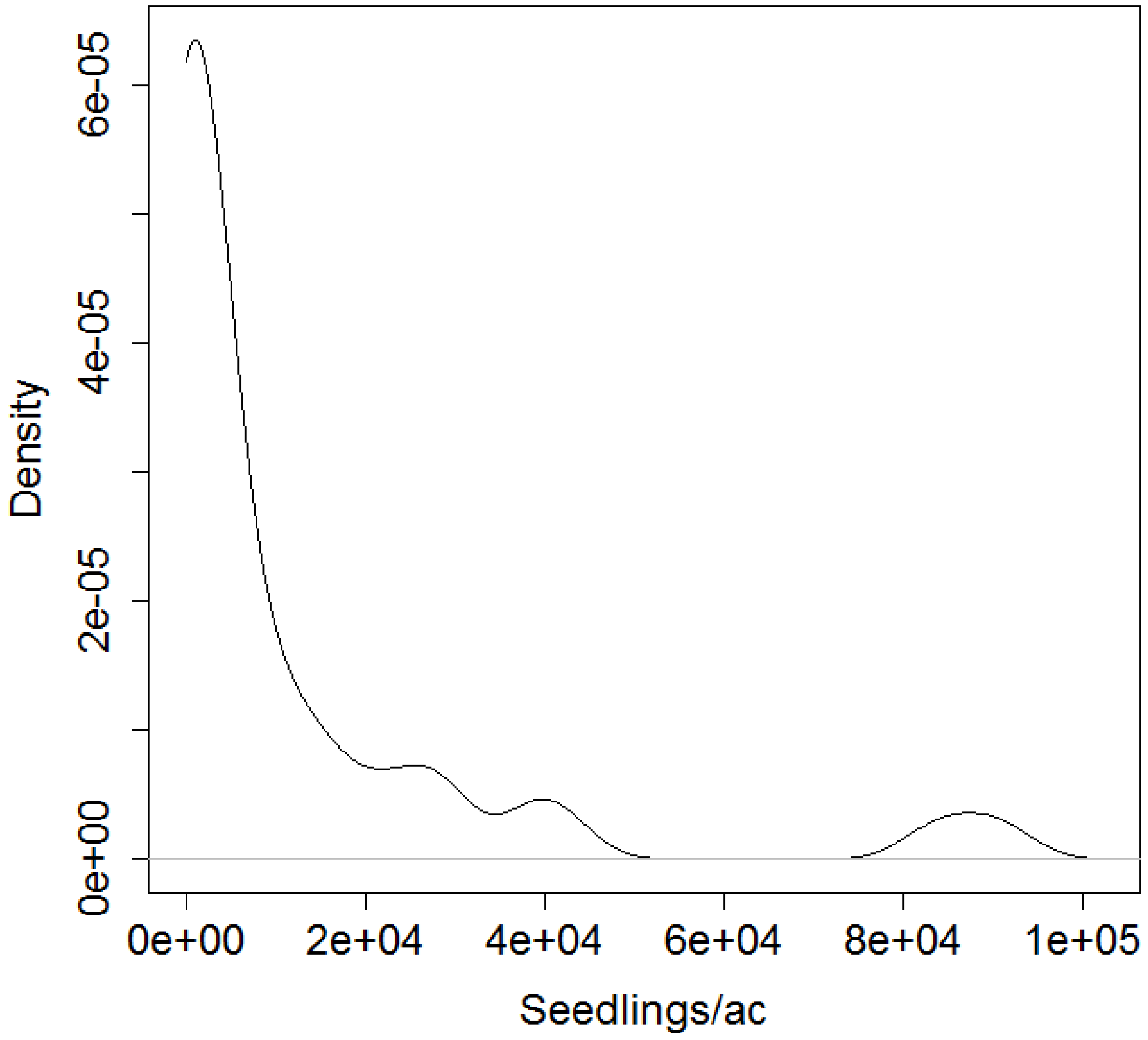


Figure 2.2. The smoothed density of seedlings per acre, showing a highly skewed distribution with lots of zeros.

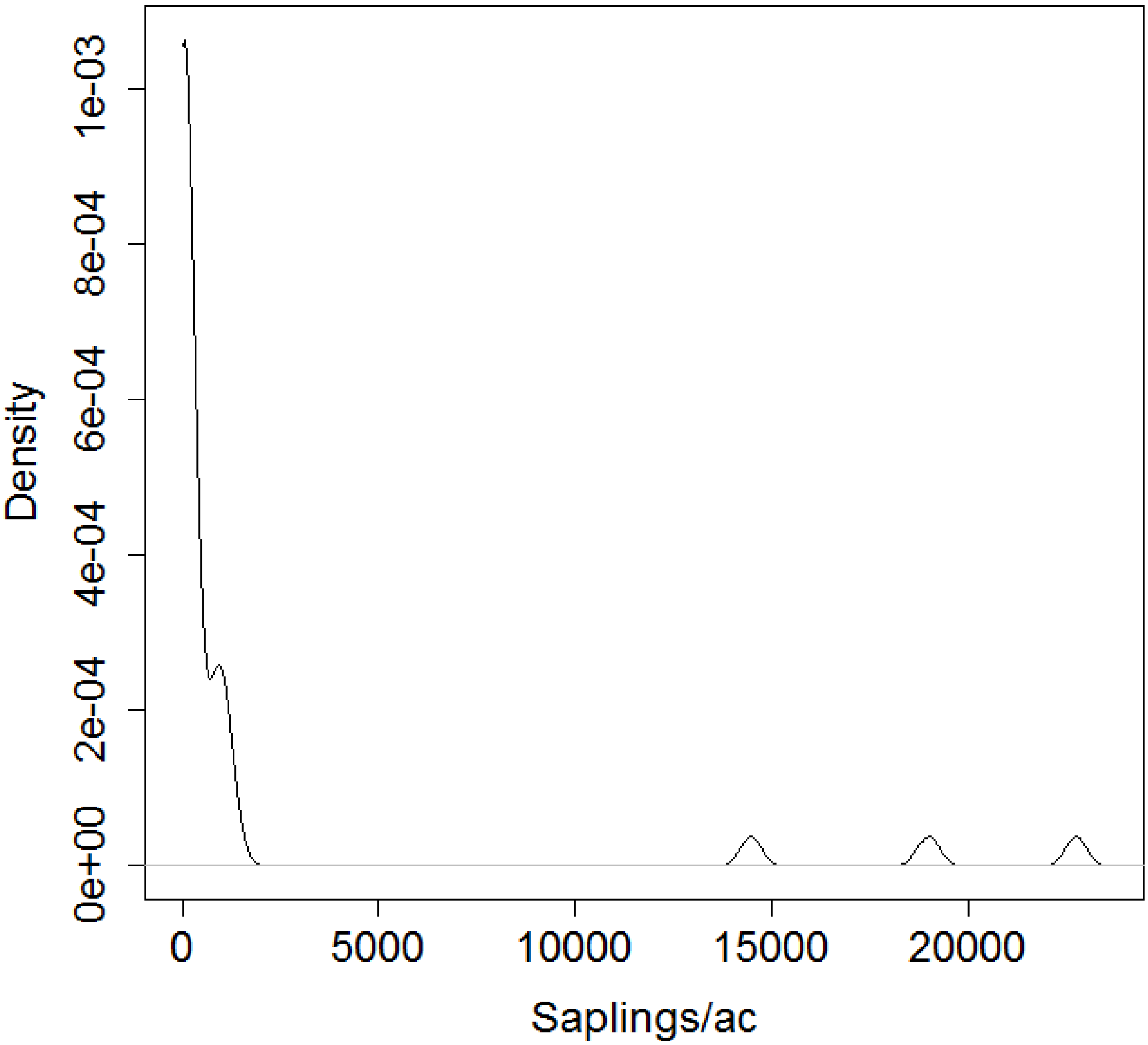


Figure 2.3. The smoothed density of saplings per acre, showing a highly skewed distribution with lots of zeros.

Zone of exclusion analysis

The following variables were used to test the hypothesis regarding the 16 meter zone of exclusion under/around mature trees: plot number, sapling height, and the distance to the nearest overstory tree. Descriptive statistics with a confidence interval equal to 95% was used.

All analyses were done using Minitab 17 Statistical Software (Minitab 2010) and R, with an alpha level of 0.05 used to determine significance. The datasets that were used for each objective are listed below.

2.3 Results

2.3.1 Seedling recruitment analysis

We converted the total number of seedlings per plot to a per acre basis. In order to determine whether there was relationship between the total number of seedlings and other factors, a Pearson's correlation was ran. Since basal area management class (P = <.001) and trees/acre (P = <.001) were both correlated with other covariates, they were removed from future regression analysis. Linear regressions were ran on the number of seedlings/acre by overstory tree age, stand basal area, time since last fire, and QMD to find possible trends within the data (Figure 2.4).

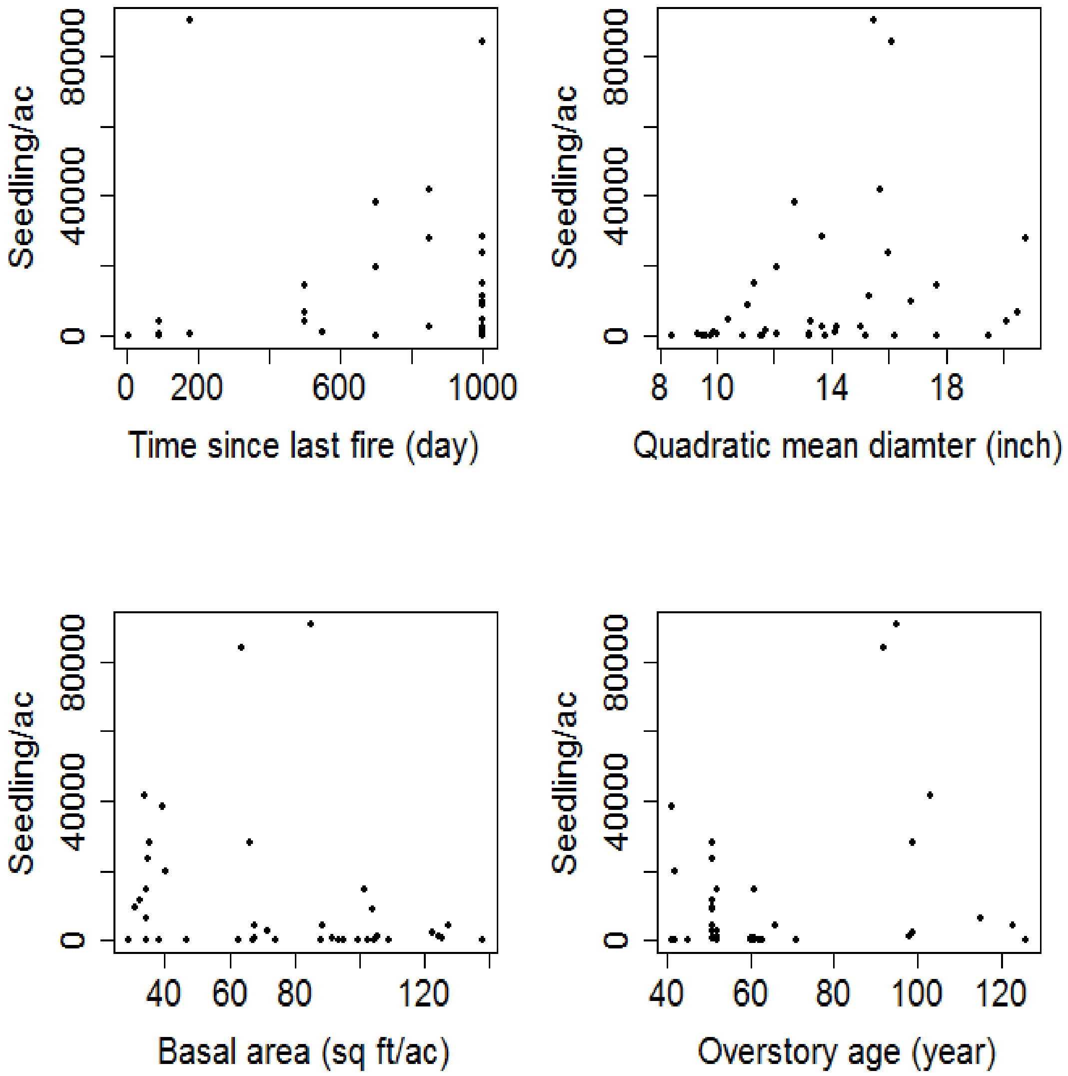


Figure 2.4. Seedlings/acre by time since last fire, QMD, basal area, and overstory tree age in the Escambia Experimental Forest. This allowed us to examine trends within the data.

Because our data was highly skewed with a large amount of ‘zeros’ within the response variable (seedlings/acre) with the variance (433066947) being significantly greater than the mean (11106.25), a zero inflated negative binomial regression was conducted. This was the best model for our data, as our AIC value was 593.07 compared to nearly 3,000 for the next nearest model. The count model results are summarized in Table 2.2. Overstory age and time since last fire were not significant in determining the total number of seedlings/acre, whereas site index and basal area were significant in determining total number of seedlings/acre. As site index goes up, the number of seedlings goes down (indicated by negative intercept). Also, as basal area increases, the number of seedlings goes down (also indicated by the negative intercept).

Table 2.2. The count model for seedlings/acre within the zero inflated negative binomial regression results are shown.

	Estimate	Std. Error	Z value	Pr(>|z|)
(Intercept)	21.335	3.561	5.991	<0.001
Overstory age	0.007	0.010	0.693	0.488
Site index	-0.156	0.042	-3.688	<0.001
Time since last fire	0.001	0.001	1.925	0.054
Basal area	-0.015	0.006	-2.344	0.019
Log(theta)	-0.033	0.243	-0.137	0.891

The binary model results are summarized in Table 2.3. All four variables (overstory age, site index, time since last fire and basal area) were significant in predicting the probability of having zero seedlings. As the overstory age, site index, and time since last fire increases, the probability of having zero seedlings decreases (interpreted from the negative estimate value). In contrast, as basal area increases, the probability of having zero seedlings increases (interpreted from the positive estimate value).

Table 2.3. The binary model for seedlings/acre within the zero inflated negative binomial regression results are shown.

	Estimate	Std. Error	Z value	Pr(>|z|)
(Intercept)	47.930	17.046	2.812	0.005
Overstory age	-0.127	0.047	-2.683	0.007
Site index	-0.551	0.200	-2.751	0.006
Time since last fire	-0.007	0.003	-2.649	0.008
Basal area	0.059	0.027	2.200	0.028

2.3.2 Seedling survival to sapling analysis

We converted the total number of saplings per plot to a per acre basis. Linear regressions were ran on the number of seedlings/acre by overstory tree age, stand basal area, time since last fire, and QMD to find possible trends within the data (Figure 2.5).

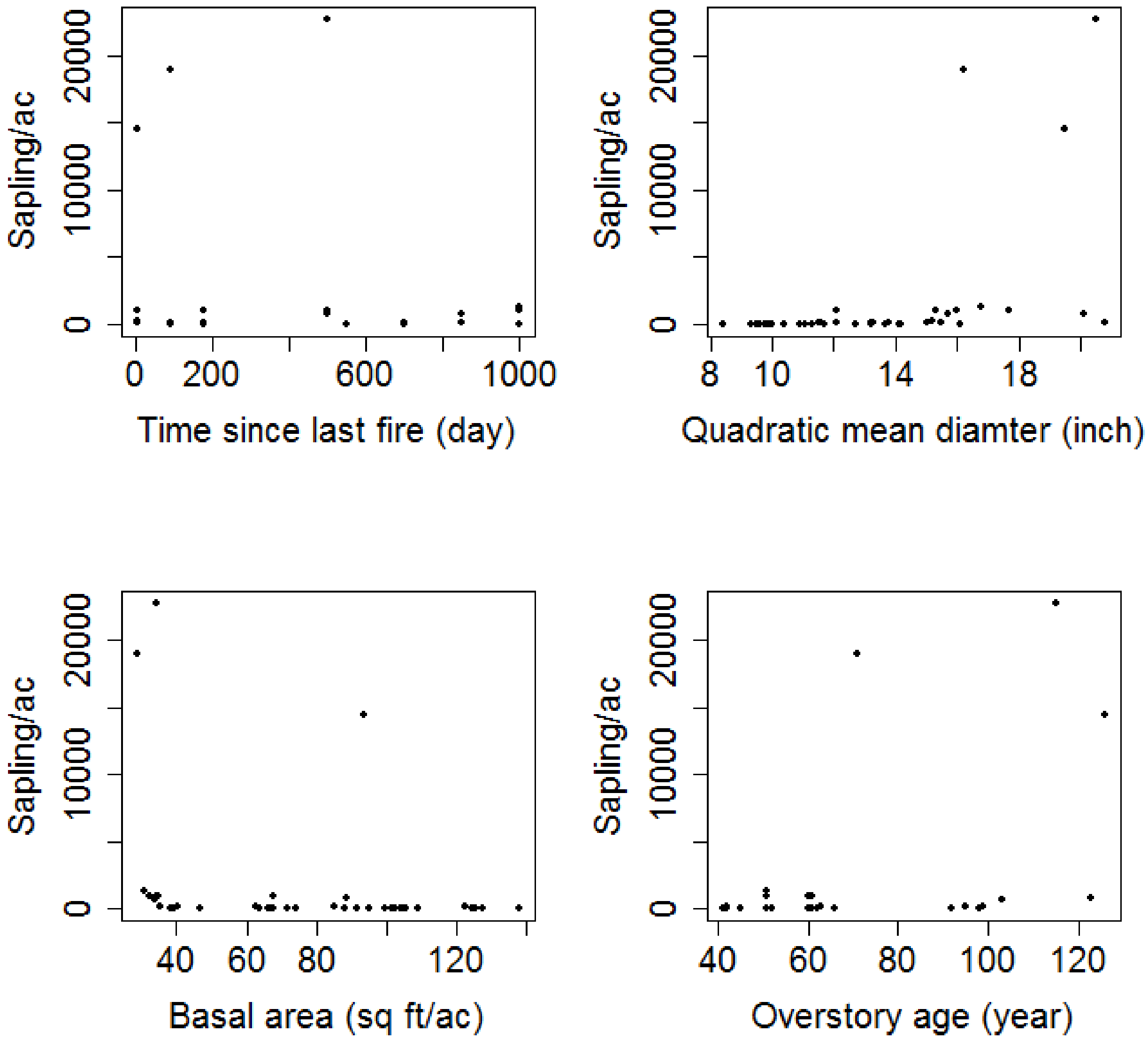


Figure 2.5. Saplings/acre by time since last fire, QMD, basal area, and overstory tree age in the Escambia Experimental Forest. This allowed us to examine trends within the data.

Because our data was highly skewed with a large amount of ‘zeros’ within the response variable (saplings/acre) with the variance (25420677) being significantly greater than the mean (1620) a zero inflated negative binomial regression was conducted. This was the best model for our data, as our AIC value was 402.11 compared to nearly 3,000 for the next nearest model. The count model results are summarized in Table 2.4. Overstory age, time since last fire, nor quadratic mean diameter were significant in determining the total number of saplings/acre. Basal area was the only significant covariate in determining saplings/acre. As basal area increases, the number of saplings decreases.

Table 2.4. The count model for saplings/acre within the zero inflated negative binomial regression results are shown.

	Estimate	Std. Error	Z value	Pr(>|z|)
(Intercept)	5.490	2.789	1.969	0.049
Overstory age	0.058	0.036	1.620	0.105
Time since last fire	-0.002	0.001	-1.828	0.068
Basal area	-0.058	0.020	-2.854	0.004
Quadratic mean diameter	-0.418	0.277	-1.506	0.132

The binary model results are summarized in Table 2.5. Quadratic mean diameter was the only significant covariate in determining the probability of having zero saplings. As QMD increases, the probability of having zero saplings decreases.

Table 2.5. The binary model for saplings/acre within the zero inflated negative binomial regression results are shown.

	Estimate	Std. Error	Z value	Pr(>|z|)
(Intercept)	13.692	7.433	1.842	0.066
Overstory age	0.112	0.065	1.720	0.086
Time since last fire	0.006	0.003	1.735	0.083
Basal area	0.001	0.026	0.030	0.976
Quadratic mean diameter	-1.955	0.892	-2.190	0.029

2.3.3 Plot 179-190 analysis

Plots 179-190 provided an opportunity to examine factors isolated from landscape level variation. These plots are a subset of plots which are all in the same management unit at the EEF. These plots all are burned at the same time, have the same age of overstory trees, and a SI_{25} 81 (+ or – 4). Basically, the only difference between the plots are distinctive different basal areas (3 at each basal area class). Running a simple linear regression for the number of seedlings / acre versus basal area yielded an R^2 = 30.32% (Figure 2.6). Running a simple linear regression for the number of saplings / acre versus basal area yielded an R^2 = 65.44% (Figure 2.7).

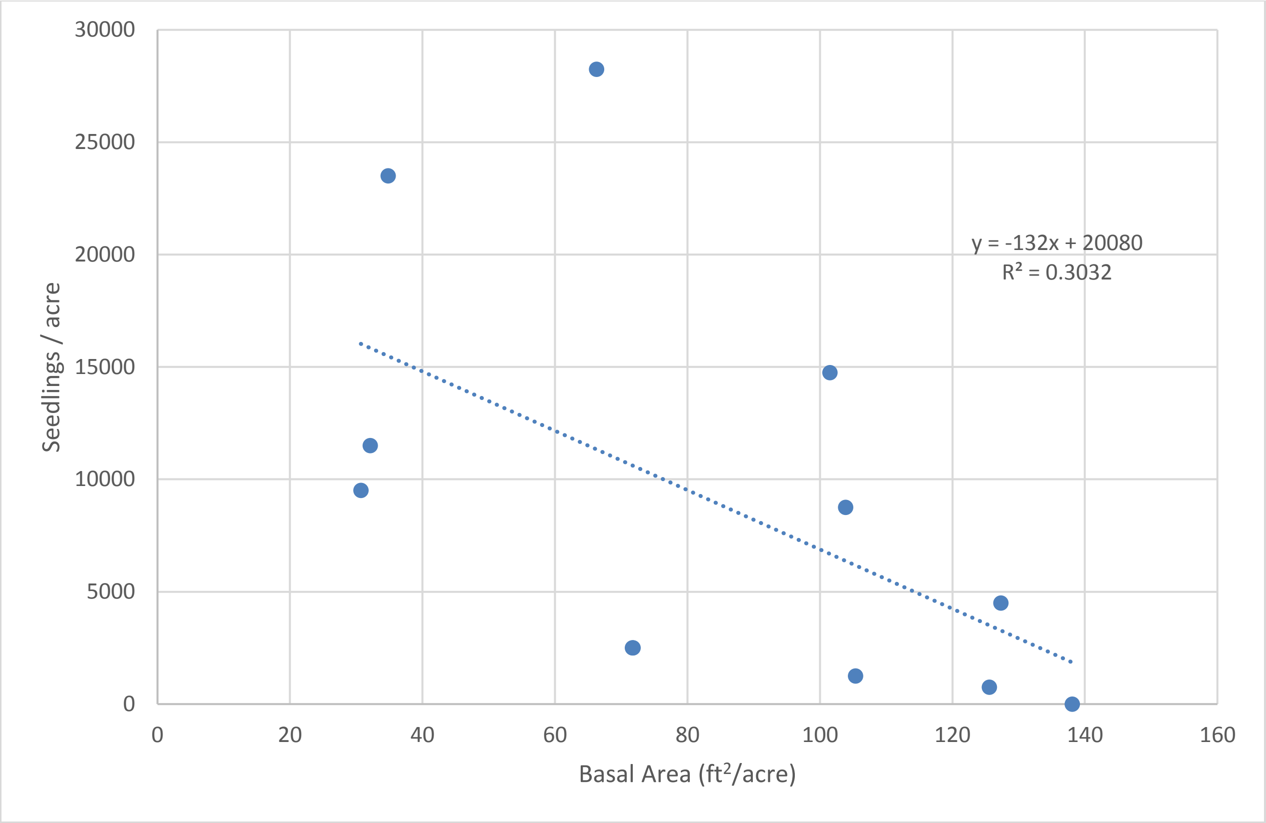


Figure 2.6. The number of seedlings / acre versus basal area is compared. For these 12 plots, basal area appears to be a significant factor in seedling establishment.

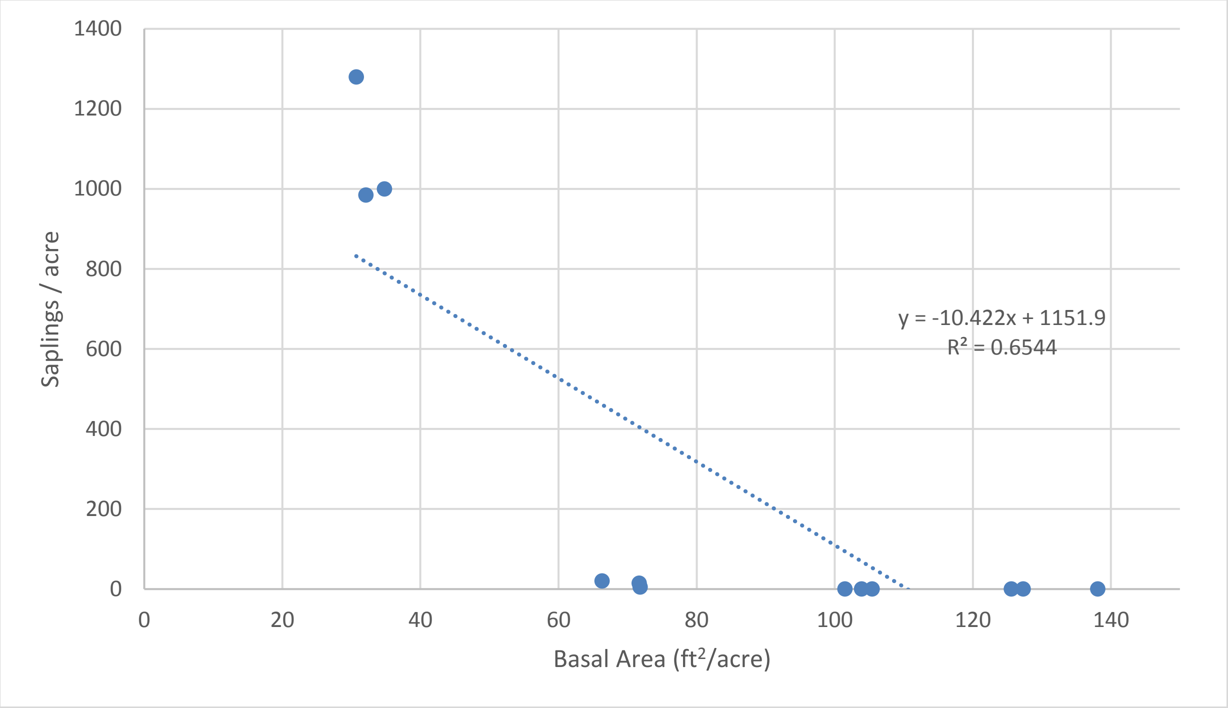


Figure 2.7. The number of saplings / acre versus basal area is compared. For these 12 plots, basal area appears to be a significant factor in sapling survival.

2.3.4 Zone of exclusion analysis

Out of the 40 plots measured in total, 15 (basal area range 39.3 – 138.1 sq. ft. / acre) did not have any saplings present. Additionally, 9 plots (basal area range 28.6 – 93.5 sq. ft. / acre) had more than 150 saplings present. Those 15 and 9 plots were excluded from analysis. Sixteen plots (basal area range 34.1 – 122.4 sq. ft. / acre) and a total of 276 saplings were used to examine the zone of exclusion. The data used for analysis included the sapling height, and the distance to the nearest overstory tree. The average distance to nearest overstory tree (zone of exclusion), was 5.32 meters (standard deviation = 2.365; Figure 2.8). There was a range of 0.270 meters (minimum) to a maximum of 13.250 meters (Figure 2.8).

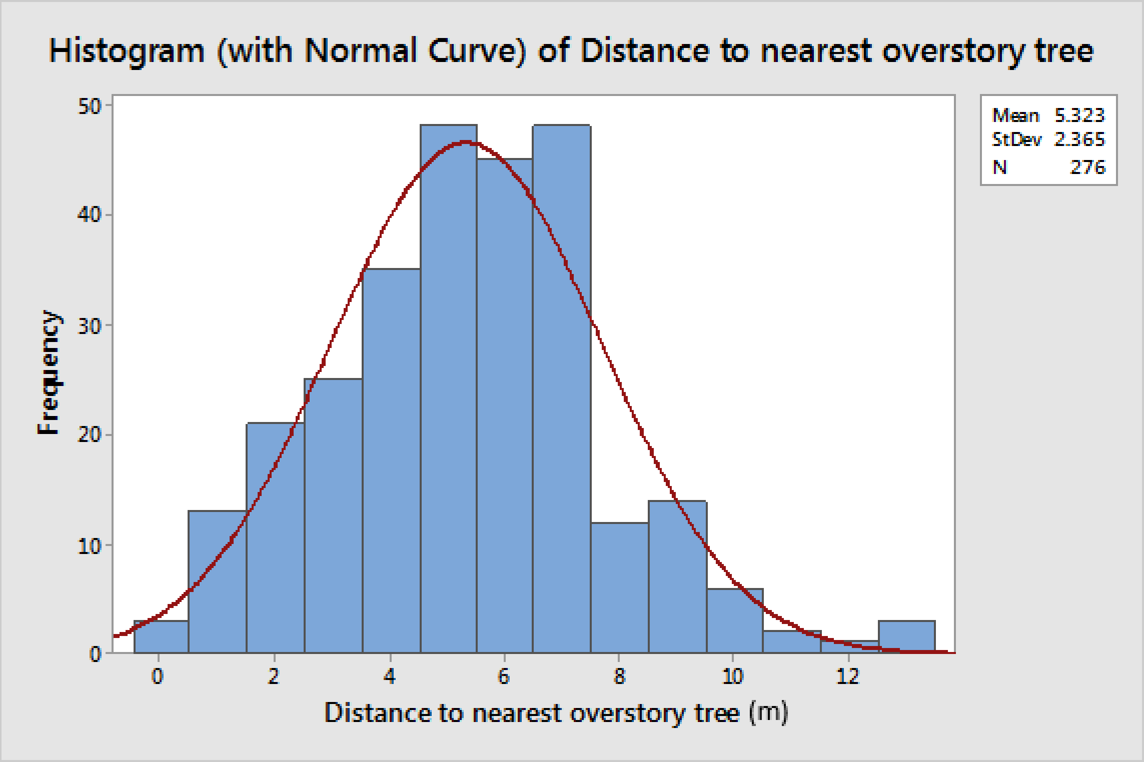


Figure 2.8. The histogram of sapling frequency with respect to distance to nearest mature tree. Normal curve is shown. The mean distance was 5.3 meters.

We also wanted to see if there was a relationship between sapling height and the distance to the nearest overstory tree. We ran a simple linear regression, and received an R^2 value of 1.87% (Figure 2.9). The result indicates that there is very minimal correlation between the size of a sapling and the distance to the nearest overstory tree.

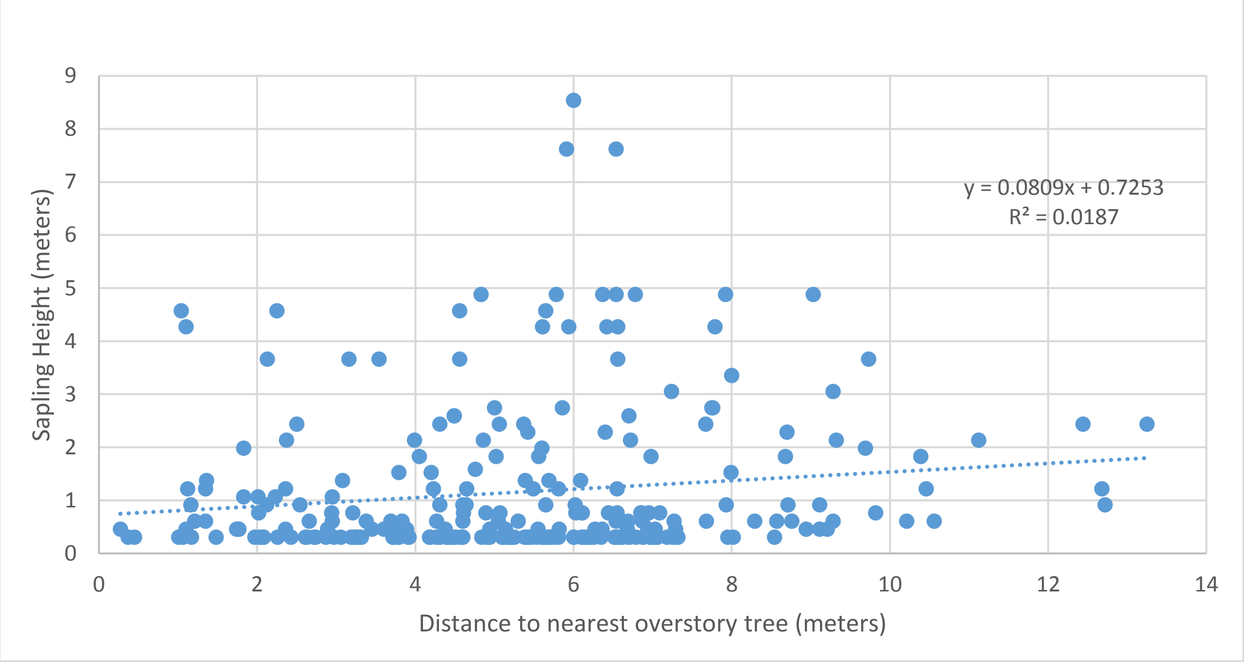


Figure 2.9. A simple linear regression was ran to see the relationship between the distance to nearest overstory tree and the height of the sapling (R^2 = 1.87%).

2.4 Discussion

2.4.1 Impact of basal area

While basal area, site index, and time since last fire were all significant indicators of seedling recruitment, basal area was the only one of those three that was also significant in determining seedling survival into future saplings. Even though basal area was significant in determining the total number of seedlings per acre (Table 2.2), we present no data to suggest that high basal area classes eliminate all possibilities of seedling recruitment. Croker (1973) also shows that although there is a decrease in seedlings with higher basal area, there are still seedlings present. Situations may arise where a land manager may not want a large amount of seedlings; therefore, thinning below 60 square feet per acre is not recommended. If a land manager wants a large amount of seedling recruitment, a heavy thinning down to near 30 square feet per acre should be achieved. Similar results can be shown on the 12 plots analyzed from the same management group (Figure 2.6). Croker (1973) show similar results that we found and suggest that your initial thinning cut should be made at least three growing seasons prior to your desired timeframe since cone crops will not be benefitted until that time (Croker and Boyer, 1975).

Once your stand has recruited new seedlings, our data suggests that you must thin to 30 square feet per acre in order for those seedlings to survive into saplings and eventual mature trees (Figure 2.7). This can also be shown from our zero inflated negative binomial regression (Table 2.4) that indicates basal area being a significant covariate in sapling survival. Croker (1973) show similar results that we found.

2.4.2 Impact of prescribed fire

Fire (whether natural or prescribed) is the single most important disturbance shaping the longleaf ecosystem (Brockway and Outcalt, 1998). Although fire did not have a significant effect on the number of seedlings or saplings/acre, it was significant in determining the probability of having no regeneration in a plot. Our data shows that as the time since fire increases, the probability of having no seedlings in a plot decreases (Table 2.3). Our observations align with the idea that longleaf pine evolved with frequent surface fires. Without these fires, hardwood tree species and other pines encroach and threaten to overtake the longleaf pine forest. These surface fires make this species habitat suitable for its own regeneration (Platt et al., 1988).

2.4.3 Zone of exclusion

Our data indicate that there is no clear zone of exclusion present in natural regeneration of longleaf pine. This is in contrast to other studies (Wahlenberg, 1946; Smith, 1955; Brockway and Outcalt, 1998) that have shown a 16 meter gap between a mature tree and seedlings. Our data is more similar to McGuire et al., (2001) that suggested mature longleaf pine overstory trees increase, rather than decrease, the establishment of longleaf pine seedlings. Although our study did not examine the relationship between overstory trees and seedlings, we are able to draw conclusions based on the fact that there were saplings present immediately under and in close proximity to the overstory trees. We can draw the conclusion that recruitment of seedlings is minimally/not affected by location to overstory trees.

2.4.4 Future studies

Since we marked the location of the seedlings on the 1/1000th acre strips, we will be able to return after a few years and examine the survivability of seedlings into saplings. This will allow land managers to make more informed decisions once they have inventoried how much regeneration they have. Depending on survivability rates, managers will then be able to identify more appropriate management techniques for this species once they do an inventory and determine how much of what is located in a stand.

2.5 Conclusions

Natural regeneration of longleaf pine should be encouraged since it does not require the purchase of seed, and costs of management are significantly lower than artificial regeneration. It also is more aesthetically pleasing and you do not lose as much wildlife as with a full harvest and artificial planting. Site index and basal area are important in determining the amount of seedlings present in a stand. In addition, basal area was the only significant indicator in determining the amount of saplings present in a stand. Fire was significant in that it had a ‘bell-curve’ effect. As time since last fire decreased, so did the amount of regeneration. Additionally, plots with 700+ days since time since last fire had minimal regeneration. It was those plots in the middle that had the most amount of regeneration, indicating a return interval of two years as optimum for maximizing regeneration in a plot. Our data indicates that there is no clear “zone of exclusion” under/around mature trees. Instead, the mature trees in our plots likely enhanced, rather than detracted, the amount of regeneration.

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