Effects of Planting Density, Prescribed Fire, and Other Factors on Stand Structure and Wildlife Habitat in Longleaf Pine Stands in Alabama

by

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Abstract

Though the distribution of longleaf pine (*Pinus palustris*) forests has decreased throughout the past century, increasing awareness of the economic and ecological benefits of longleaf silviculture has stimulated restoration efforts via plantation forestry. However, designing planting and management prescriptions that effectively balance wildlife habitat and timber production objectives in plantations can be difficult. While planting to greater densities may increase potential revenues, some wildlife-focused restoration programs implement planting density restrictions due to the concern that densely planted stands will reduce the amount and duration of availability of herbaceous understory vegetation, negatively impacting wildlife habitat quality. However, the outcomes of these restrictions and the influence of prescribed fire in mitigating density concerns have not been thoroughly evaluated. Therefore, we initiated a study to examine the contributions of planting density and management history on stand structure and understory vegetation in select pre-commercial thin longleaf stands in the Coastal Plain of Alabama.

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CHAPTER 1

EFFECTS OF PLANTING DENSITY, PRESCRIBED FIRE, AND OTHER FACTORS ON UNDERSTORY VEGETATION AND ASSOCIATED WILDLIFE HABITAT IN LONGLEAF PINE STANDS

ABSTRACT

Though longleaf pine (*Pinus palustris*) forest coverage has declined drastically, increasing awareness of the economic and ecological benefits of longleaf silviculture has increased interest in restoration, especially via plantation forestry. The primary objective of such programs is often to enhance or restore habitat for wildlife dependent on herbaceous understory plant communities. Because herbaceous cover is often inversely related to canopy cover, restoration programs often place restrictions on longleaf planting density. However, the influence of planting density on understory plant communities has not been thoroughly evaluated. Therefore, we initiated a study to examine the relative influences of planting density, stand age, current basal area, and prescribed fire on understory composition and associated wildlife habitat in select pre-commercial thin longleaf stands in the Coastal Plain of Alabama during 2017–2018. We did not detect an effect of planting density, prescribed fire return, or stand age on understory vegetation. However, for each 1 m^2/ha increase in current longleaf basal area, coverage of herbaceous and woody plants decreased by 3.5% and 2.4%, respectively. Coverage of preferred northern bobwhite (*Colinus virginianus*) forage plants similarly decreased 1.9% for each 1 m^2/ha increase in longleaf basal area. None of the factors we evaluated were significant predictors of white-tailed deer (*Odocoileus virginianus*) forage. Our findings related to planting density were likely a

function of low longleaf pine survival, which is not uncommon. Because of this and the inherent variability in growth rates for young longleaf pine stands, restoration programs should perhaps place greater emphasis on post-planting monitoring and management. Additionally, our results related to northern bobwhite and white-tailed deer forage demonstrate that longleaf pine restoration does not automatically create high quality habitat for all species.

INTRODUCTION

Though longleaf pine (*Pinus palustris*) forests once covered as much as 37 million ha of the southeastern United States, coverage was reduced to approximately 3.5% (1.3 million ha) of its original extent by 1995 (Landers et al., 1995). Longleaf pine forests are still valued for providing wildlife habitat, supporting high levels of floristic diversity, and producing valuable timber products (Hedman et al., 2000). Reasons for the decline in longleaf include unsustainable logging, conversion of lands to other uses or faster growing pine species (e.g., loblolly pine [*P. taeda*]), and fire suppression (Landers et al., 1995; Outcalt, 2000; Stainback and Alavalapati, 2004). The decline in longleaf coverage has resulted in a significant decline in associated flora and fauna and, as of 2006, 66% of species classified as declining, threatened, or endangered in the southeastern United States were associated with the longleaf ecosystem (Mitchell et al., 2006).

However, increasing awareness of both the economic and ecological benefits of longleaf pine has increased interest in restoration. The major emphasis of these efforts centers around the ecological benefits associated with promoting an herbaceous understory. Concomitantly, financial assistance programs have encouraged longleaf pine restoration on private lands, and plantation forestry has been proposed as a viable means of wide-scale longleaf restoration (Harrington, 2011). For example, the Natural Resources Conservation Service (NRCS) offers longleaf planting assistance to private landowners through the Environmental Quality Incentives Program (NRCS, 2017).

When converting or restoring land to longleaf pine, planting density is one of the primary factors that will affect stand structure and associated wildlife habitat quality during the short- and long-term. From a silvicultural perspective, planting density influences stocking rates, planting costs, wood quality and volume, and timing of operational factors such as harvesting and thinning (Huang et al., 2005). Greater planting densities provide a buffer against seedling mortality associated with competition and herbivory, and increase potential timber revenues (Harrington, 2011). Further, Albritton (2012) suggested that greater planting densities (e.g., ≥1,483 seedlings/ha) allow stands to reach canopy closure and naturally prune lower limbs sooner, resulting in a greater number of high-quality trees. Others (e.g., Demers et al., 2000) have recommended longleaf pine planting densities ≥1,854 seedlings/ha if timber production is a primary objective.

However, because canopy closure occurs sooner in densely planted stands, coverage of understory vegetation and duration of availability will decline earlier in the life of the stand, impacting wildlife habitat (Harrington, 2006). Specifically, denser stands compete with understory vegetation through the combined effects of overstory shading, needle-fall, and belowground competition (Harrington, 2011). Therefore, wildlife-focused longleaf pine restoration generally calls for decreased planting densities. For example, Demers et al. (2000) recommended planting from 742–1,236 seedlings/ha if the goal is longleaf ecosystem restoration and/or wildlife habitat. Further, South (2006) suggested that a planting density of 1,100 seedlings/ha would be more optimal for producing herbivore forage than a density of 2,200 seedlings/ha.

However, prescribed fire is also an important driver of wildlife habitat quality in longleaf pine forests. Coupled with the more open canopy associated with this species, prescribed fire encourages a species-rich herbaceous understory (Van Lear et al., 2005). For example, frequent (i.e., every 1–3 years), low-intensity fire limits invasive plant coverage, prepares the seedbed for natural longleaf regeneration, increases understory plant diversity, and stimulates seed production of other native species (Frost, 1993; Aschenbach et al., 2009). In the absence of frequent fire, woody shrubs and trees will eventually develop a midstory, suppressing herbaceous plant coverage by shading plants near ground level (Kush et al., 1999; Loudermilk et al., 2011). In contrast, frequent fire in longleaf stands can result in some of the most species-rich plant communities outside of the tropics (Hedman et al., 2000), and plant species densities as great as 42/0.25 m^2 have been recorded in longleaf pine savannas (Drew et al., 1998). Therefore, even in relatively low-density longleaf stands, absence of frequent prescribed fire may preclude occupancy, or limit abundance, of focal wildlife species.

Nonetheless, wildlife-focused longleaf restoration programs generally place greater emphasis on planting densities than prescribed fire. For example, planting densities are restricted to a range of 989–1,691 trees/ha under EQIP in Alabama (NRCS, 2014). Anecdotally, these guidelines may be overly restrictive as abundant herbaceous vegetation may be maintained, even in densely planted stands, when frequent prescribed fire is applied. If this is the case, some longleaf restoration programs may be unnecessarily restrictive, decreasing landowner participation and ultimately limiting longleaf restoration efforts. However, research on longleaf planting regimes and the associated wildlife habitat is limited (Harrington, 2011).

Therefore, we initiated a study to examine the relative influences of planting density, prescribed fire, and other factors on stand structure and understory composition in plantation longleaf stands throughout the Coastal Plain of Alabama. We also examined how these factors affected coverage of important seed and forage plants for northern bobwhite (*Colinus virginianus*) and white-tailed deer (*Odocoileus virginianus*), two important game species in the region. We hypothesized that coverage of understory vegetation (including forage plants) would be inversely related to planting density. We also hypothesized that coverage of herbaceous plants, northern bobwhite forage, and white-tailed deer forage would increase with increasing fire frequency, and that coverage of woody vegetation would decrease with increasing fire frequency.

STUDY AREAS

We conducted our study in 9 pre-commercial thin longleaf pine stands on private lands in the Coastal Plain of Alabama (Figure 1.1). All stands were ≥5 years old, planted to a specific density (i.e., not regenerated naturally), prepared for planting via broadcast herbicide application, and ≥4 ha in area. Stand 1 was in Escambia County and had Orangeburg fine sandy loam and Benndale-Orangeburg complex soils. Stand 2 was in Conecuh County and had Greenville sandy loam and Troup-Orangeburg association soils. Stand 3 was in Barbour County and had Luverne sandy loam, Goldsboro loamy fine sand, Mantachie, Kinston, and Iuka soils. Stand 4 was in Bullock County and had Conecuh sandy loam soils. Stand 5 was in Macon County and had Bonifay loamy fine sand and Lucy-Luverne complex soils. Stand 6 was in Barbour County and had Luverne-Springhill complex and Conecuh sandy loam soils. Stand 7 was in Lowndes County and had Nankin-Springhill-Lucy complex, Cowarts sandyloam, Bonifay loamy sand, and Lucy loamy sand soils. Stand 8 was in Lowndes County and had Nankin-Springhill-Lucy complex and Bonifay loamy sand soils. Stand 9 was in Barbour County and had Luverne sandy loam, Troup-Alaga complex, Mantachie, Kinston, Iuka, and Luverne-Springhill complex soils (NRCS, 2019).

The study region generally had hot summers, mild winters, and year-round precipitation. Specifically, daytime high summer temperatures typically ranged from 29–35 °C, average winter low temperatures ranged from -1–7 °C, and average annual statewide precipitation totals were 137 cm (Runkle et al., 2017).

METHODS

Prior to sampling, we mapped stand boundaries in ArcMap10.4.1 (Environmental Systems Research Institute, Inc., Redlands, CA). We subset stands >8 ha into 8-ha units and randomly selected one unit for sampling using a random number generator in program R (R Core Team, 2019). We used a fishnet grid to systematically locate a series of points spaced 50-m apart within each stand and randomly selected sample points from the grid at a density of 1/0.4 ha. Points were distributed proportionately (based on area) between interior (>50 m from boundary) and edge (≤50 m from boundary) portions of the stand. We performed vegetation sampling at each point during the summers of 2017 and 2018. Specifically, we established a 30-m transect along a random azimuth originating at each point and identified each species of plant that intersected the transect at 3-m increments (10 total points) according to the FIREMON Point Intercept Sampling Method (Caratti, 2006). When multiple plants of the same species intersected a single point, we recorded a hit for each independent plant; therefore, it was possible to have total cover values >100% for a transect.

For each transect, we calculated the percent cover of herbaceous (i.e., grasses and forbs) and woody plants (i.e., trees, shrubs, and woody vines). We also calculated the percent cover of plants considered moderate to highly preferred white-tailed deer (*Odocoileus virginianus*) forage based on available literature (Warren and Hurst, 1981; Miller and Miller, 1999; Table A1). We did the same for plants considered valuable seed and soft mast producers for northern bobwhite

(*Colinus virginianus*), according to the literature (Landers and Johnson, 1976; Rosene and Freeman, 1988; Miller and Miller, 1999; Table A2).

Additionally, from the set of points used for establishing vegetation sampling transects, we selected timber cruise points at a density of 1/0.8 ha and a distance of 100 m apart. Points were distributed proportionately (based on area) between interior (>50 m from boundary) and edge (≤50 m from boundary) portions of the stand. We conducted a timber cruise at each point during January–March 2018. Specifically, we established a 30-m transect along a random azimuth originating at each point and measured diameter at breast height (DBH) of all longleaf pines ≥1.4 m in height within ≤5 m of the transect, such that area sampled constituted an approximately 5% cruise. For each cruise point we calculated the current density (trees/ha), basal area (m^2/ha), and average DBH for longleaf pine. We defined planting density for each stand as the target longleaf pine planting density based on information provided by landowners, agency officials, and land managers.

We used linear regression in program R (R Core Team, 2019) to estimate the effects of stand-level parameters (i.e., longleaf pine planting density, stand age, and average prescribed fire return interval [stand age ÷ number of prescribed fires]) on percent cover of each category of plants. We compared stand-level models using AICc (Burnham and Anderson, 2004), and considered those within ≤2 ΔAICc points of the top model competitive. We generated parameter estimates and associated 95% confidence intervals for each fixed effect parameter in each competitive model. We considered parameters with 95% confidence intervals not overlapping zero informative (Arnold, 2010), and set α=0.05 for all tests.

We also constructed linear mixed-effects models in the nlme package (Pinheiro et al., 2018) to estimate the effect of longleaf basal area on our response variables for transects where we collected both timber and vegetation data. To account for the structure of our data, sample point was nested within stand as a random effect in each model. We used the effects package (Fox et al., 2018) to construct plots of the estimated effects of each significant predictor from informative models on response variables of interest. In cases where the null model was the only competitive model, we concluded that we did not have evidence of an effect of any of the predictors in our data set.

RESULTS

We collected data from a total of nine stands that met our criteria. Stand size ranged from 5–8 ha and planting density ranged from 1,078–1,538 longleaf seedlings/ha. Percent of longleaf target planting density remaining in stands (current longleaf density ÷ longleaf planting density) averaged 46%. During 2017, stand age ranged from 6–16 years, and average fire return interval ranged from 2–7 years (Table 1.1). Although some of our stand-level predictors of interest were contained in the competing model set for stand-level response variables, none of these predictors were informative (i.e., their confidence intervals overlapped zero; Table 1.2).

In contrast, longleaf pine basal area was a significant predictor of percent cover of herbaceous and woody plants, as well as northern bobwhite forage plants. Specifically, for each 1 m^2/ha increase in longleaf basal area, percent cover of herbaceous plants decreased 3.5%, percent cover of woody plants decreased 2.3%, and percent cover of northern bobwhite forage plants decreased 1.9% (Table 1.3, Figure 1.2).

DISCUSSION

Although the role of planting density in shaping longleaf stands and understory plant communities has been heavily debated, and longleaf pine planting densities are restricted under financial assistance programs such as EQIP (NRCS, 2014), we did not detect an effect of planting density on herbaceous plant coverage. We believe this finding is likely attributable to post-planting mortality. Specifically, longleaf density in our stands was, on average, only 46% of the original planting density. Others have reported similar findings for longleaf stands in a variety of age classes. For example, Knapp et al. (2006) reported longleaf survival was as low as 57% 20 months post-planting for plots prepared with herbicide, and Knapp et al. (2015) reported longleaf survival in clearcuts was 40% at the end of the fifth growing season. In addition, Jack et al. (2010) measured survival of longleaf seedlings for two years after application of prescribed fire and found that survival was only 30–50%, depending on the season of burn. Finally, South et al. (2012) reported that average survival across a number of studies was 51% for plantation longleaf stands ranging in age from 10–28 years old. Although our study design did not allow us to directly determine the factors that contributed to low survival, planting density was not a good predictor of current density and, by extension, a poor predictor of canopy cover, which could interfere with understory vegetation development.

However, we did find that increasing longleaf basal area had a negative effect on herbaceous and woody cover, as well as coverage of seed and soft mast producers used by northern bobwhite. In general, this is consistent with much of the literature. For example, in a study of young plantation longleaf and slash pine (*P. elliottii*), biomass of herbaceous plants decreased 73 kg/ha for each 1 m^2/ha increase in pine basal area (Wolters, 1973). Similarly, Harrington and Edwards (1999) reported a 21% increase in herbaceous coverage in response to thinning young longleaf plantations from 9 m^2/ha to 5 m^2/ha. Specifically regarding northern bobwhite habitat requirements, Stransky (1971) recommended a maximum longleaf pine basal area of 14 m^2/ha, and Little et al. (2009) suggested pine basal area should not exceed 9 m^2/ha when northern bobwhite habitat is a primary management objective. However, Wolters (1982) found that coverage of herbaceous vegetation in longleaf stands was not significantly impacted until 17 years post-planting. In contrast, our oldest stand was 17 years old in 2018, and the average age of our stands was 10 years in 2017 and 11 years in 2018. Further, Wolters (1981) suggested that maintaining a pine basal area of 12–20 m^2/ha and applying prescribed fire, both conditions which were met in our study, could sustain an herbaceous-dominated understory.

Given the importance of light availability in herbaceous production (Pecot et al., 2007), it is possible that the influence of basal area on understory vegetation may vary with the age and size of trees, attributable to differential light attenuation. For example, Gaines et al. (1954) suggested that there may be an upward trend of herbaceous production in older stands with higher basal area, but fewer trees and an increase in side light due to taller trees. These concerns may be especially important for plantation stands, given that rates of crown closure often exceed those of natural stands (Harrington, 2006). Therefore, it is important for managers to be aware that the negative influences of increasing basal area may become apparent sooner in planted versus naturally regenerated longleaf stands.

It is well-established that prescribed fire promotes herbaceous vegetation by reducing litter accumulation, reducing or removing competing vegetation, and influencing overstory structure (Boyer, 1990; Boyer, 1993; Harrington and Edwards, 1999), and the influence of average fire return interval on herbaceous vegetation in our study approached statistical significance. Fire frequency is more important than season in maintaining understory vegetation structure consistent with longleaf ecosystem restoration objectives (Addington et al., 2015; Glitzenstein et al., 2008). However, a 1–3 year fire return is necessary for limiting woody plant abundance in longleaf understories (Addington et al., 2015), and Glitzenstein et al. (2003) suggested that even slight reductions in fire frequency can stimulate sprouting and proliferation of shrubs, reduce space available for herbaceous plants, and decrease species richness. Therefore, the absence of a statistically significant effect of fire on woody vegetation in our study may be attributable to the fact that the average fire return interval in our stands was 3.5 years and 3.7 years during 2017 and 2018, respectively.

Similarly, we did not detect an influence of prescribed fire return on northern bobwhite forage plants. One potential reason is that the majority of these plants are promoted by growing season fire, whereas the majority of prescribed fire events on our sites occurred during the dormant season. Specifically, early growing season fire promotes both native warm season grasses and forbs, whereas late growing season fire may promote additional forb coverage and decrease woody encroachment (Harper, 2007). Further, Haywood (2009) found that month of burning significantly affected herbaceous plant cover in young longleaf stands, with July-burn plots having significantly greater grass and forb cover than March-burn or May-burn plots. In addition to the potential influences of season of burn, our stands were exposed to prescribed fire less frequently than is generally recommended (i.e., 2 years; Burke et al., 2008) for northern bobwhite habitat management. Regardless, our finding does not necessarily imply that our stands were not suitable for northern bobwhite. Specifically, although we did not evaluate cover, it has been established that native, perennial grasses, which were abundant in our stands, are important for northern bobwhite nesting material and cover (Greenfield et al., 2002). Therefore, our stands may have provided adequate nesting and predator concealment cover for this species.

None of the factors we evaluated significantly influenced coverage of preferred white-tailed deer forage plants. This is not surprising given that most preferred deer forage plants are either forbs or woody browse species (Warren and Hurst, 1981; Miller and Miller, 1999). Many of our sites were dominated by grasses, which have little to no food value for deer, and may preclude more preferred forb species (Felix III et al., 1986). In addition, the increased coverage of woody browse species in the less frequently burned stands may have been counteracted by the increased coverage of herbaceous plants in more-frequently burned stands. Therefore, deer forage was spread out between more and less frequently burned stands, and there was a lack of a detectable directional effect of fire on overall deer forage availability.

These findings have important implications for advancing our understanding of the primary drivers of understory structure and associated wildlife habitat quality in young plantation longleaf pine stands. Specifically, the absence of any detectable effects of planting density on understory responses of interest, combined with previously observed variation in growth and survival common among young longleaf plantations, suggests that post-planting monitoring and management guidelines may be more important than those related to planting density for government subsidized longleaf restoration programs. Specifically, given the observed negative influence of increasing longleaf basal area on herbaceous, woody, and preferred northern bobwhite forage plants, it may be important for managers to actively monitor basal area and understory vegetation and use thinning and prescribed fire to maintain preferred understory conditions, as necessary. In addition, although relatively high coverage of herbaceous vegetation generally benefits a number of longleaf-associated wildlife species, it is important to be aware that high herbaceous coverage does not necessarily provide optimal habitat for all species throughout the year. Rather, stand management, including management actions like prescribed fire, should be prescribed and periodically evaluated on a case-by-case basis to ensure habitat conditions for focal species are being met (Harper, 2007).

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TABLES AND FIGURES

Table 1.1. Management history for longleaf pine (*Pinus palustris*) stands in the Coastal Plain of Alabama where we evaluated the effects of planting density and stand management on coverage of herbaceous and woody plants, white-tailed deer (*Odocoileus virginianus*) forage, and seed and soft mast producers for northern bobwhite (*Colinus virginianus*) during 2017–2018.

Stand	**Area (ha)**	**Planting Density (trees/ha)**	**Age**[a]	**Average Fire Return**[b]
1	5	1,078	8	2.7
2	8	1,122	10	3.3
3	8	1,197	8	2.7
4	7	1,345	12	2.0
5	8	1,345	14	7.0
6	8	1,347	16	4.0
7	8	1,360	14	3.5
8	7	1,483	6	3.0
9	8	1,538	6	3.0

[a] Stand age (years) in summer 2017

[b] Stand age ÷ number of prescribed fires

Table 1.2. Number of parameters (K), Akaike's Information Criterion (AICc), difference from lowest AICc (ΔAICc), and model weights (w_i) for competitive models used to predict the effects of longleaf pine (*Pinus palustris*) planting density and stand management on percent cover of herbaceous and woody plants, white-tailed deer (*Odocoileus virginianus*) forage, and seed and soft mast producers for northern bobwhite (*Colinus virginianus*). Stands were located in the Coastal Plain of Alabama and sampled during 2017–2018.

Model	**K**	**AICc**	**ΔAICc**	w_i
Herbaceous				
Age[a]	3	17.2	0.00	0.23
Fire Return[b]	3	17.3	0.04	0.22
Null	2	17.3	0.12	0.21
Fire Return + Planting Density[c]	4	19.1	1.90	0.09
Woody				
Age	3	9.8	0.00	0.43
Null	2	11.2	1.44	0.21
Preferred deer forage				
Age	3	17.6	0.00	0.39
Null	2	18.2	0.60	0.29
Northern bobwhite forage				
Null	2	-12.8	0.00	0.37
Fire Return	3	-12.1	0.69	0.26
Age	3	-10.8	1.91	0.14

[a] Stand age (years) for the vegetation sampling year

[b] Stand age ÷ number of prescribed fires

[c] Longleaf pine planting density (seedlings/ha)

Table 1.3. Parameter estimates (β), standard errors (SE), lower (LCL) and upper (UCL) 95% confidence limits, and *P*-values (α = 0.05) for informative parameters from models used to predict the effects of longleaf pine (*Pinus palustris*) basal area on percent cover of herbaceous and woody plants, white-tailed deer (*Odocoileus virginianus*) forage, and seed and soft mast producers for northern bobwhite (*Colinus virginianus*) for stands in the Coastal Plain of Alabama sampled during 2017–2018.

Model	**β**	**SE**	**LCL**	**UCL**	***P*-value**
Herbaceous					
Intercept	149.90	13.17	123.88	175.93	<0.001
Basal Area[a]	-3.45	1.38	-6.18	-0.73	0.014
Woody					
Intercept	77.57	10.61	56.60	98.55	<0.001
Basal Area	-2.28	1.14	-4.54	-0.02	0.049
Preferred deer forage					
Intercept	128.71	14.08	100.88	156.54	<0.001
Basal Area	-2.37	1.49	-5.33	0.59	0.118
Northern bobwhite forage					
Intercept	52.41	6.57	39.42	65.40	<0.001
Basal Area	-1.87	0.71	-3.28	-0.47	0.010

[a] Longleaf pine basal area (m^2/ha)

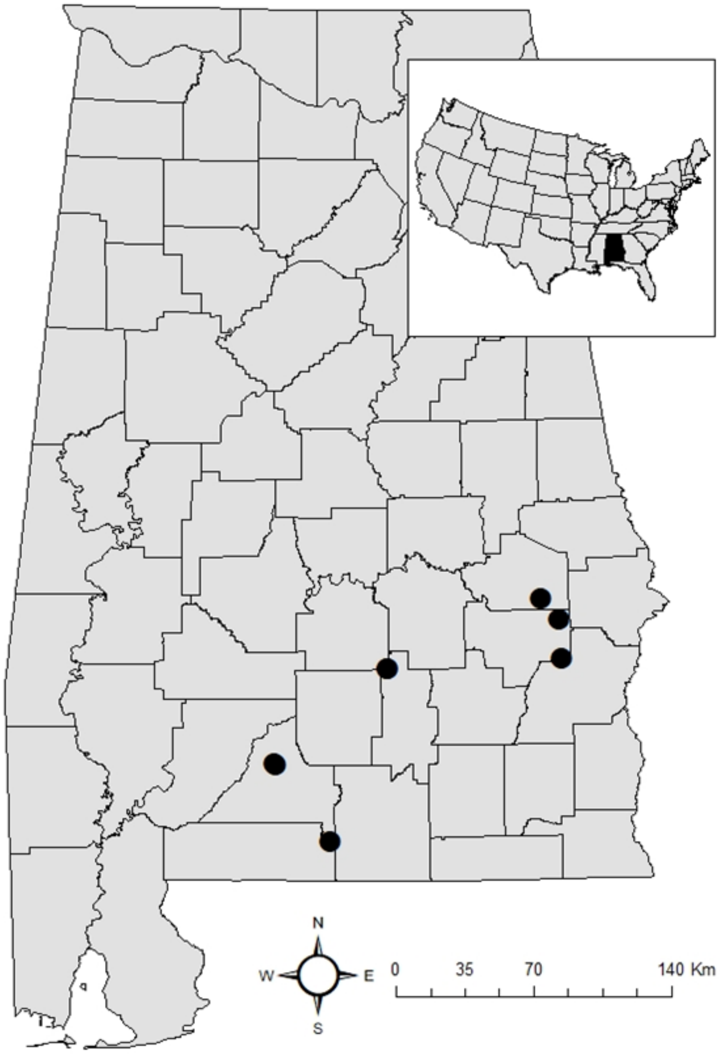

Figure 1.1. General locations of longleaf pine (*Pinus palustris*) stands in the Coastal Plain of Alabama where we evaluated the effects of longleaf pine planting density and stand management on coverage of herbaceous and woody plants, white-tailed deer (*Odocoileus virginianus*) forage plants, and valuable seed and soft mast producers for northern bobwhite (*Colinus virginianus*) during 2017–2018.

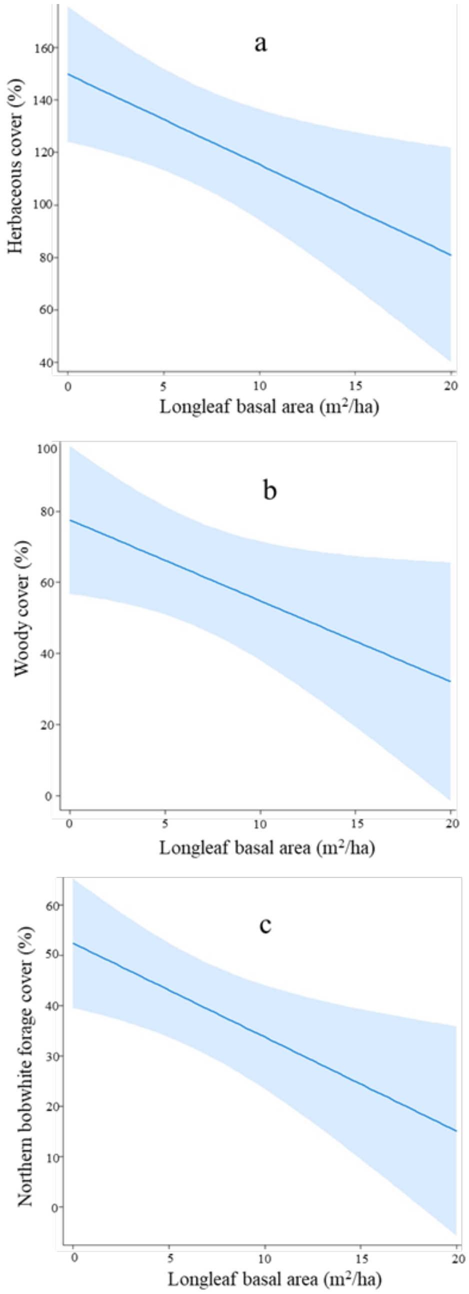

Figure 1.2. Plots predicting the effects of longleaf pine (*Pinus palustris*) basal area on the understory percent cover of (a) herbaceous plants, (b) woody plants, and (c) plants valuable as seed and soft mast producers for northern bobwhite (*Colinus virginianus*) for longleaf pine stands in the Coastal Plain of Alabama sampled during 2017–2018.

CHAPTER 2

EFFECTS OF PLANTING DENSITY AND PRESCRIBED FIRE ON LONGLEAF PINE STAND CHARACTERISITCS AND STRUCTURE

ABSTRACT

Although the major emphasis of longleaf pine (*Pinus palustris*) restoration efforts centers on the benefits associated with longleaf ecosystems, it can also be an economically viable timber species. Plantation forestry has been proposed as a means of longleaf restoration, but wildlife-focused landowner incentive programs often restrict planting density. However, there is uncertainty whether the planting density restrictions associated with restoration programs impede timber production objectives, and how other management alternatives (e.g., prescribed fire) affect stand structure and other characteristics. Therefore, we initiated a study to examine the relative influences of planting density and prescribed fire on stand structure in select pre-commercial thin plantation longleaf stands throughout the Coastal Plain of Alabama during 2017–2018. Percent of target planting density remaining in stands varied from 31–64%, but neither prescribed fire nor the other factors we examined were significant predictors of current density. Our findings also are in agreement with others who have suggested that planting to greater densities is warranted if a well-stocked stand of high quality trees is a landowner objective.

INTRODUCTION

Although the major emphasis of longleaf pine (*Pinus palustris*) restoration efforts centers on the benefits associated with longleaf ecosystems, it can also be an economically rewarding timber species. For example, longleaf pine produces a wide range of timber products that garner premium prices, and produces more product dry weight per unit volume than other southern pines (Landers et al., 1995). From a stand management perspective, longleaf pines are also more fire tolerant and resistant to fusiform rust and pests such as bark beetles (Landers et al., 1995; Alavalapati et al., 2002). Although longleaf is generally considered to be one of the slower growing southern pine species, longleaf growth rates can be increased via cultivation in plantations (Kush et al., 2006).

Concomitant with increasing awareness of both the economic and ecological benefits of longleaf pine, financial assistance programs have been implemented to encourage restoration on private lands, and plantation forestry has been proposed as a viable means of widespread longleaf restoration (Harrington, 2011). However, landowner incentive programs are generally wildlife focused and, because wildlife habitat quality declines sooner in densely planted stands (Harrington, 2006), these programs often restrict planting density. For example, the Natural Resources Conservation Service (NRCS) offers longleaf planting assistance to private landowners through the Environmental Quality Incentives Program (EQIP), but limits planting densities to a range of 989–1,691 trees/ha in Alabama (NRCS, 2014).

Silviculturally, planting density influences planting costs, stocking rates, wood quality and volume, and timing of operations such as thinning and final harvest (Huang et al., 2005). Planting to greater densities provides a buffer against seedling mortality and allows stands to reach canopy closure and naturally prune lower limbs sooner, resulting in a greater number of high-quality trees and increased revenue (Harrington, 2011; Albritton, 2012). Accordingly, Demers et al. (2000) recommended longleaf pine planting densities ≥1,854 seedlings/ha if timber production is a primary objective, which is beyond the range allowed under most longleaf-associated EQIP programs.

However, prescribed fire also affects timber and wildlife habitat objectives within longleaf stands. From a timber perspective, prescribed fire is largely responsible for the competitive success of longleaf pine. Specifically, longleaf pines are intolerant of competition and fire plays a major role in reducing competition with grass-stage seedlings and increasing survival (Boyer, 1993; Gilliam and Platt, 1999; Haywood, 2000). Additionally, fire encourages height growth and reduces risk of brown-spot needle blight (Boyer and Peterson, 1983). In the absence of frequent fire, woody shrubs and trees will eventually form a midstory, and long-term fire suppression may completely preclude formation of a longleaf canopy (Kush et al., 1999; Kush, 2016). From an ecosystem perspective, frequent fire in longleaf stands can result in some of the most species-rich plant communities outside of the tropics (Hedman et al., 2000), which provide valuable wildlife habitat.

Therefore, it is possible that timber and wildlife objectives are not mutually exclusive in longleaf stands. However, others have documented negative effects of frequent prescribed fire on longleaf pine stands (Boyer, 1993; Haywood, 2000), and there is uncertainty whether the planting density restrictions associated with longleaf restoration programs impede timber production objectives (Harrington, 2011). If so, more timber-minded landowners may choose not to participate in programs, limiting longleaf restoration efforts.

Therefore, we initiated a study to examine the relative influences of planting density and prescribed fire on stand structure in plantation longleaf stands throughout the Coastal Plain of Alabama. Specifically, we examined how these factors affected current longleaf density, basal area, and non-longleaf basal area. We predicted that greater planting densities would positively influence longleaf basal area, while we were uncertain of the directional effect of prescribed fire on current density and basal area due to inconsistencies in the literature. Finally, we predicted that non-longleaf basal area would decrease as fire frequency increased due to the competitive advantage of longleaf pine in frequently burned systems.

STUDY AREAS

We conducted our study in 9 pre-commercial thin longleaf pine stands on private lands in the Coastal Plain of Alabama (Figure 2.1). All stands were ≥5 years old, planted to a specific density (i.e., not regenerated naturally) on sites that were prepared via broadcast herbicide application, and ≥4 ha in area. Stand 1 was in Escambia County and had Orangeburg fine sandy loam and Benndale-Orangeburg complex soils. Stand 2 was in Conecuh County and had Greenville sandy loam and Troup-Orangeburg association soils. Stand 3 was in Barbour County and had Luverne sandy loam, Goldsboro loamy fine sand, Mantachie, Kinston, and Iuka soils. Stand 4 was in Bullock County and had Conecuh sandy loam soils. Stand 5 was in Macon County and had Bonifay loamy fine sand and Lucy-Luverne complex soils. Stand 6 was in Barbour County and had Luverne-Springhill complex and Conecuh sandy loam soils. Stand 7 was in Lowndes County and had Nankin-Springhill-Lucy complex, Cowarts sandy loam, Bonifay loamy sand, and Lucy loamy sand soils. Stand 8 was in Lowndes County and had Nankin-Springhill-Lucy complex and Bonifay loamy sand soils. Stand 9 was in Barbour County and had Luverne sandy loam, Troup-Alaga complex, Mantachie, Kinston, Iuka, and Luverne-Springhill complex soils (NRCS, 2019). The study region generally had hot summers, mild winters, and year-round precipitation. Specifically, daytime high summer temperatures typically ranged from 29–35 °C, average winter low temperatures ranged from -1–7 °C, and average annual statewide precipitation totals were 137 cm (Runkle et al., 2017).

METHODS

Prior to sampling, we mapped stand boundaries in ArcMap10.4.1 (Environmental Systems Research Institute, Inc., Redlands, CA). We subset stands >8 ha into 8-ha units and randomly selected one unit for sampling using a random number generator in program R (R Core Team, 2019). We used a fishnet grid to systematically locate a series of points spaced 100-m apart within each stand and randomly selected cruise points from the grid at a density of 1/0.8 ha. Points were distributed proportionately (based on area) between interior (>50 m from boundary) and edge (≤50 m from boundary) portions of the stand. We conducted a timber cruise at each point during January–March 2018. Specifically, we established a 30-m transect along a random azimuth originating at each point and measured diameter at breast height (DBH) of all longleaf pines ≥1.4 m in height within ≤5 m of the transect, such that area sampled constituted an approximately 5% cruise. Additionally, we recorded species and DBH of all non-longleaf pine woody vegetation ≥7.6 cm DBH within cruise plots. For each cruise point we calculated the current density (trees/ha), basal area (m^2/ha), and average DBH for longleaf pine. We defined planting density for each stand as the target longleaf pine planting density based on information provided by landowners, agency officials, and land managers. For non-longleaf pine woody vegetation, we calculated the current density (stems/ha), basal area, and average DBH.

We used simple linear regression in program R (R Core Team, 2019) to estimate the effects of stand-level parameters (i.e., longleaf pine planting density and average prescribed fire return interval [stand age ÷ number of prescribed fires]) on current longleaf density, current longleaf basal area, and current non-longleaf basal area. We generated parameter estimates and associated 95% confidence intervals for each fixed effect parameter in each competitive model. We considered parameters with 95% confidence intervals not overlapping zero informative

(Arnold, 2010), and set $\alpha=0.05$ for all tests. In cases where the null model was the only competitive model, we concluded that we did not have evidence of an effect of any of the predictors in our data set.

RESULTS

We collected data from a total of nine stands that met our criteria. Stand size ranged from 5–8 ha, age ranged from 8–16 years, planting density ranged from 1,078–1,538 seedlings/ha, and average fire return interval ranged from 2.2–7.5 years (Table 2.1). Current longleaf density ranged from 334–855 trees/ha. Although there were two competing models describing the relationship between stand-level factors and current density, the intercept only (i.e., null) model carried the most weight and the parameter estimate for fire return was uninformative (Table 2.2). Longleaf basal areas ranged from 1–15 m^2/ha. Although there were two competing models describing the relationship between stand-level factors and longleaf basal area, the intercept only (i.e., null) model carried the most weight and the parameter estimate for fire return was uninformative (Table 2.2). Non-longleaf basal area ranged from 0–9 m^2/ha. However, the intercept only (i.e., null) model was the only competitive model describing the effects of this factor, leading to the conclusion that none of our parameters were informative predictors of non-longleaf basal area in the stands we sampled (Table 2.2).

DISCUSSION

We did not find longleaf planting density to be a significant predictor of current density and current densities in our stands averaged only 46% of the target planting density. Others have reported similar results. For example, Haywood (2007) found that longleaf seedling survival was as low as 52% in unburned plots after six growing seasons, and noted that survival decreased with age. Further, longleaf survival was as low as 60% only 2–4 years post-establishment in intensively managed plantations (Dyson and Brockway, 2015). However, others have reported significantly greater longleaf survival rates. For example, Hu et al. (2012) reported seedling survival in clearcuts as high as 80% after three growing seasons and Cram et al. (2010) reported 87% survival in a 15-year-old plantation in South Carolina.

Mortality in young longleaf stands is commonly attributed to a variety of factors, including fire, drought, and competition from non-longleaf vegetation (Haywood, 2000; Haywood, 2007). Although our study design did not allow for direct determination of survival rates or mortality sources, current density was not associated with average fire return interval on our study sites. This may have resulted from our limited sample size. However, it is important that the potential effect of fire on current density was not of sufficient magnitude to be detected in our study given that frequent prescribed fire is critical for maintaining the desired conditions associated with longleaf restoration objectives (Van Lear et al., 2005), and many managers are concerned with fire-associated mortality. Additionally, although fire can kill young longleaf pine trees, it is important to remember that prescribed fire can also benefit longleaf stands by relieving grass-stage seedlings from the stresses of competition with woody and herbaceous vegetation, as well as brown-spot needle blight (Haywood, 2000).

Additionally, we did not detect an effect of fire on longleaf basal area. In contrast, others have attributed reductions in growth rates to fire. Specifically, Boyer (1993) found that biennial fires reduce growth rates in young longleaf stands, regardless of season of burn, accounting for approximately 75% of observed differences in volume growth when compared to unburned stands. However, in that study fire effects diminished with time and were undetectable by age 24.

Although we did not detect any effects of prescribed fire on stand structure, it is important to point out that fire-suppression in longleaf forests may also result in undesirable outcomes. Specifically, fire suppression leads to encroachment of various hardwood species, as well as other southern yellow pine species that threaten longleaf recruitment and slow growth (Brockway and Lewis, 1997; Provencher et al., 2001; Shappell and Koontz, 2015). The absence of a detectable effect of fire on non-longleaf basal area was likely related to the timing of fire in our stands. Specifically, if fire was not implemented early enough in the rotation, the faster growing non-longleaf species may have reached sufficient size to become less susceptible to fire. For example, we only included trees ≥7.6 cm DBH in our calculation of non-longleaf basal area and most of these were loblolly pine, which are significantly less vulnerable to fire once they are ≥4 cm DBH (McNabb, 1977). Given the known importance of fire in longleaf silviculture, it is also important to consider that fire may have had more of an effect on stand structure than our data suggest, but that we were not able to detect it statistically due to the broad range of fire return intervals and limited number of stands included in our study.

Nonetheless, the absence of a correlation between planting density and current density, in combination with a relatively low percent of planted trees remaining, suggests that planting to greater densities is warranted if a well-stocked stand of high quality trees is a landowner objective. Specifically, planting to greater densities benefits stand management by providing a buffer against seedling mortality (Harrington, 2011), and may result in a greater number of high-quality trees (Albritton, 2012), increasing timber revenues. However, balancing commodity production with ecological values is one of the greatest challenges facing forest managers (Hedman et al., 2000).

As a result, longleaf restoration incentive programs such as EQIP, which restricts planting densities to 989–1,691 trees/ha in Alabama (NRCS, 2014), often limit planting densities. In our stands, this would result in only 465–795 trees/ha several years post-planting, which is significantly less than the recommendation for stands where timber production is a primary objective (Demers et al., 2000). Therefore, such restrictive programs may limit participation in landowner incentive programs and, therefore, overall longleaf restoration efforts. Instead, and especially given the variability in survival and growth rates within young longleaf pine stands, we suggest such programs place greater emphasis on post-establishment monitoring, particularly longleaf basal area and its effects on herbaceous vegetation in light of our separate findings, when wildlife habitat is an objective.

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TABLES AND FIGURES

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Stand	**Area (ha)**	**Planting Density (trees/ha)**	**Age**[a]	**Average Fire Return**[b]
1	5	1,078	9	3.0
2	8	1,122	11	3.7
3	8	1,197	9	3.0
4	7	1,345	13	2.2
5	8	1,345	15	7.5
6	8	1,347	17	3.4
7	8	1,360	15	3.8
8	7	1,483	7	3.5
9	8	1,538	7	3.5

[a] Stand age (years) in January 2018

[b] Stand age ÷ number of prescribed fires

Table 2.2. Number of parameters (K), Akaike's Information Criterion (AICc), difference from lowest AICc (ΔAICc), and model weights (w_i) for models used to predict the effects of longleaf pine (*Pinus palustris*) planting density and stand management on stand condition for stands in the Coastal Plain of Alabama sampled during January–March 2018.

Model	**K**	**AICc**	**ΔAICc**	w_i
Longleaf density[a]				
Null	2	121.6	0.00	0.63
Fire Return[b]	3	123.2	1.65	0.27
Planting Density[c]	3	125.5	3.92	0.09
Planting Density + Fire Return	4	129.7	8.09	0.01
Longleaf basal area				
Null	2	61.0	0.00	0.50
Fire Return	3	61.4	0.36	0.43
Planting Density	3	65.5	4.45	0.06
Planting Density + Fire Return	4	68.4	7.35	0.01
Non-longleaf basal area				
Null	2	47.7	0.00	0.84
Fire Return	3	52.3	4.64	0.08
Planting Density	3	52.5	4.79	0.07
Planting Density + Fire Return	4	59.5	11.83	0.01

[a] Longleaf pine density (trees/ha)

[b] Stand age ÷ number of prescribed fires

[c] Longleaf pine planting density (seedlings/ha)

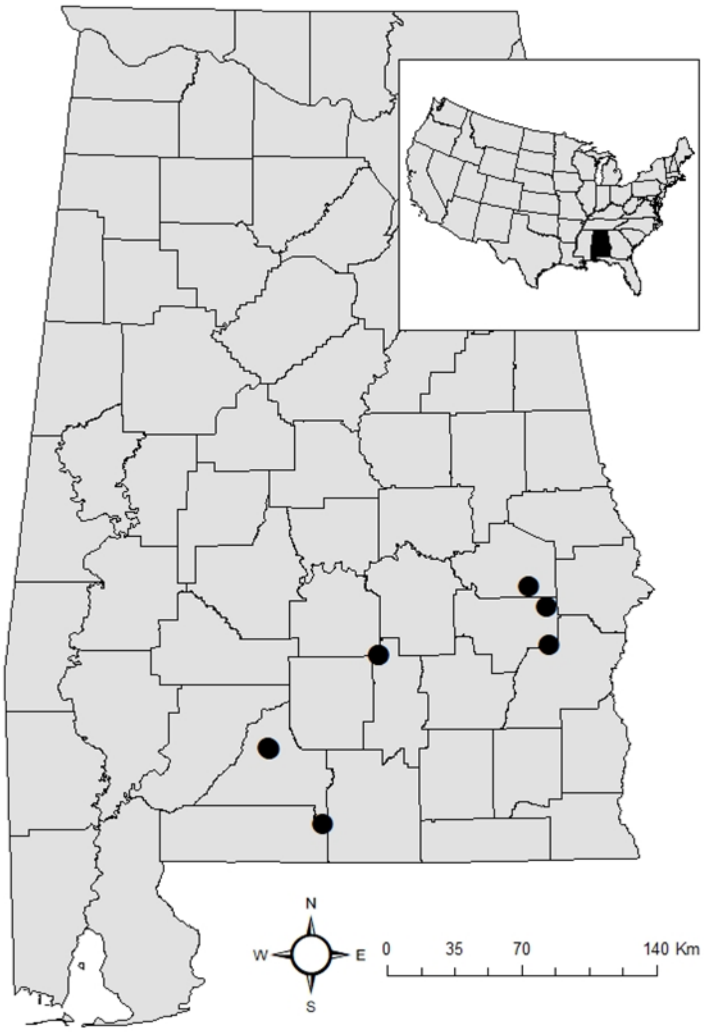

Figure 2.1. General locations of longleaf pine (*Pinus palustris*) stands in the Coastal Plain of Alabama where we evaluated the effects of longleaf pine planting density and stand management on stand condition during January–March 2018.

APPENDIX

Table A1. List of plant species considered moderate to highly preferred white-tailed deer (*Odocoileus virginianus*) forage from vegetation data collected in 9 longleaf pine (*Pinus palustris*) stands in the Coastal Plain of Alabama during 2017–2018.

Scientific name	**Growth habit**	**Scientific name**	**Growth habit**
Acalypha spp.	Non-legume Forb	*Lonicera japonica*	Vine
Acer rubrum	Tree	*Lysimachia quadrifolia*	Non-legume Forb
Ageratina spp.	Non-legume Forb	*Muhlenbergia schreberi*	Grass
Ambrosia artemisiifolia	Non-legume Forb	*Panicum anceps*	Grass
Aralia spinosa	Shrub	*Polypremum procumbens*	Non-legume Forb
Baccharis halmifolia	Shrub	*Prunus* spp.	Tree
Callicarpa americana	Shrub	*Rhexia* spp.	Non-legume Forb
Campsis radicans	Vine	*Rhus glabra*	Shrub
Centrosema virginianum	Legume Forb	*Rhynchosia* spp.	Legume Forb
Chamaecrista spp.	Legume Forb	*Robinia* spp.	Tree
Clitoria mariana	Legume Forb	*Rubus* spp.	Brambles
Conzya canadensis	Non-legume Forb	*Ruellia* spp.	Non-legume Forb
Coreopsis major	Non-legume Forb	*Salix* spp.	Tree
Desmodium spp.	Legume Forb	*Sanicula canadensis*	Non-legume Forb
Diodia spp.	Non-legume Forb	*Sassafras albidum*	Tree
Erigeron spp.	Non-legume Forb	*Smilax* spp.	Vine
Eupatorium spp.	Non-legume Forb	*Solidago* spp.	Non-legume Forb
Fraxinus spp.	Tree	*Stylosanthes biflora*	Legume Forb
Galactia spp.	Legume Forb	*Toxicodendron radicans*	Semiwoody
Galium spp.	Non-legume Forb	*Tragia* spp.	Non-legume Forb
Helenium amarum	Non-legume Forb	*Trichostema dichotomum*	Non-legume Forb
Helianthus spp.	Non-legume Forb	*Ulmus* spp.	Tree
Hypericum spp.	Semiwoody	*Vaccinium* spp.	Shrub
Ilex spp.	Tree	*Verbena* spp.	Non-legume Forb
Lactuca spp.	Non-legume Forb	*Vicia* spp.	Legume Forb
[*]*Lespedeza* spp.	Legume Forb	*Vitis* spp.	Vine
Liatris spp.	Non-legume Forb	*Zanthoxylum americanum*	Tree
Liriodendron tulipifera	Tree		

[*] Does not include exotic *Lespedeza* spp.

Table A2. List of species and growth habits for plants identified as valuable seed and soft mast producers for northern bobwhite (*Colinus virginianus*) from vegetation data collected in 9 longleaf pine (*Pinus palustris*) stands in the Coastal Plain of Alabama during 2017–2018.

Scientific name	**Growth habit**
Acalypha spp.	Non-legume Forb
Ambrosia artemisiifolia	Non-legume Forb
Callicarpa americana	Shrub
Campsis radicans	Vine
Carex spp.	Sedge
Centrosema virginianum	Legume Forb
Chamaecrista spp.	Legume Forb
Clitoria mariana	Legume Forb
Desmodium spp.	Legume Forb
Diodia spp.	Non-legume Forb
Galactia spp.	Legume Forb
Helianthus spp.	Non-legume Forb
Hypericum spp.	Semiwoody
Ilex spp.	Tree
Ipomoea spp.	Vine
Jacquemontia tamnifolia	Vine
[*]*Lespedeza* spp.	Legume Forb
Panicum spp.	Grass
Trichostema dichotomum	Non-legume Forb
Vaccinium spp.	Shrub
Vicia spp.	Legume Forb

[*] Does not include exotic *Lespedeza* spp.