Summer 8-5-2016

LONG-TERM OVERSTORY VEGETATION RESPONSES TO PRESCRIBED FIRE MANAGEMENT FOR LONGLEAF PINE AT BIG THICKET NATIONAL PRESERVE

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By

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Presented to the Faculty of the Graduate School of

Stephen F. Austin State University

In Partial Fulfillment

Of the Requirements

For the Degree of

Master of Science

STEPHEN F. AUSTIN STATE UNIVERSITY

August, 2016

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ABSTRACT

At the western edge of the longleaf pine (*Pinus palustris*) range, federal land managers have burned the forests of Big Thicket National Preserve to bring back the structure and diversity of the longleaf pine forest. In the early 1990's, a four year study was conducted by Rice University, and the National Park Service continued monitoring the study's fire ecology research plots. After two decades of data collection, ordination was applied to species abundance data to examine changes in vegetation communities from a variety of prescribed fire treatments and controls. The vegetation data was separated by size class to include overstory, small tree, sapling, and seedling data. Across the size classes and treatments, the sandhill and wetland savanna vegetation types remained less effected by fire treatments and only the upland pine responded to changes in the overstory. Although fire management had an effect on vegetation types, upon reviewing prescribed histories, it became evident that prescribed fire alone was not changing vegetation communities to foster longleaf pine habitat. Most of the plots did not have longleaf pine trees or seedlings present and two plots that were mechanically treated showed distinction among other treatment regimes. Restoration treatments including the mechanical and chemical application and seedling plantings are necessary to ensure restoration of the longleaf pine forest structure and diverse understory vegetation.

ACKNOWLEDGEMENTS

I would like to thank my thesis advisor, Dr. Brian Oswald, of the College of Forestry and Agriculture at Stephen F. Austin State University. Dr. Oswald had extensive patience and support as I balanced a full time job, family life, and a graduate degree. I benefited from his understanding of fire management on federal lands and knowledge of longleaf pine ecosystems.

I am also grateful to the National Park Service and countless Big Thicket National Preserve employees who worked with me. For 12 years, I was inspired by their dedication and commitment to the natural resources of the Big Thicket and I learned a great deal about restoration from working beside them. I also want to recognize past researchers, most notably Dr. Paul Harcombe, who set a foundation for research and provided valuable information for the National Park Service.

Finally, I must express my very profound gratitude to my husband Bob for providing me with continuous encouragement during my years of study and throughout the process of writing this thesis. This accomplishment would not have been possible without him.

Thank you,

Deanna Boensch

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INTRODUCTION

After European settlement, old-growth longleaf pine (*Pinus palustris* Mill.) forests were cleared faster than the current rate of deforestation among temperate and tropical rainforests (Simberloff 1993). In the moist tropical rainforests of South America, Africa, and Asia, forty percent of the forests have been deforested (Simberloff 1993); however, over the past four hundred years, ninety-seven percent of the longleaf pine ecosystem was converted for commercial logging, agriculture development, and boxed for turpentine (Frost 1993). The longleaf pine forests once dominated the southeastern United States across nine states from Virginia to Texas, with a range of 29-38 million hectares (Frost 1993, Harcombe et al. 1993, Barnett 1999, Smith 2001). The most critical element in the longleaf pine ecosystem is the occurrence of fire (Outcalt 2001), which fosters a dense and diverse herbaceous layer with up to 300 species per hectare, many of which are currently rare or endangered (Means 1996).

Historically, lightning and Native Americans periodically set fires that formed the open, grassy, park-like forest structure of the longleaf community. Prior to fragmentation of the ecosystem, only weather and topography regulated the boundaries of a fire (Johnson and Gjerstad 1998). Grassy herbaceous vegetation fueled low to medium intensity surface fires, which occurred every two to ten years during the growing season (Robbins and Myers 1992, Goebel et al.

2001, Outcalt 2001). Dendrochronological studies in Louisiana examined fire scars in remnant longleaf pines and calculated a 2.2 year mean fire return interval from 1650-1905, with a range from 0.5-12 years (Stambaugh et al. 2011). In the early 1900's, aggressive fire suppression tactics changed the "common and unregulated" fire regime characteristic of the southeast, to allow for forest protection and pine regeneration (Waldrop et al. 1992). Fire exclusion in the longleaf pine forests has been one of the most important factors in the decline of the ecosystem (Frost 1993, Hermann 2001). Without fire, other pines, hardwoods, and brush species start to occupy the midstory and out-compete native longleaf seedlings (Heyward 1939, Landers et al. 1995).

Managers currently focused on longleaf pine restoration must choose best management strategies for preservation and restoration of the ecosystem. Therefore, it is critical to understand the effects of fire in order to create prescriptions which determine the frequency, intensity, and seasonality of the burns. Glitzenstein et al. (1995) concluded intensity and fire behavior as the most important factor that accounts for species composition in the longleaf pine ecosystem, rather than seasonality and frequency. Waldrop et al. (1992) point out that higher intensity fires, such as non-prescribed wildfires, would result in a higher mortality of invading pine and hardwoods. The intensity and fire behavior are very important to consider, especially with a majority of the longleaf ecosystem in an altered state. The accumulation of dead and down woody fuels and the presence of dense understory brush and saplings as a result of fire suppression could work to either the advantage or disadvantage of longleaf pine. In these forests, fire alone may not alter the composition, since higher intensity fires have resulted in a greater density of brush resprouts than the pre-burn conditions (Hodgkins 1958). Therefore, managers will often use mechanical and chemical treatments to reduce fuel loadings and brush density.

Fire frequency has a strong effect on ground cover vegetation with annual or biennial return intervals resulting in high species diversity; furthermore, seasonality did account for differences in fire effects, but had a weaker influence (Streng and Harcombe 1982). Composites and some legumes responded better with August versus January burning (Hodgkins 1958). In contrast, Drewa et al. (2002) found that dormant season fires have greater stem densities of hardwood resprouts compared to growing season fires. Hodgkins (1958) observed more shrub and woody vine resprouts from January burns in comparison to August burns, and Waldrop et al. (1992) found that summer fires maintain fire dependent grasslands without fostering regeneration of hardwoods, where winter fires only regulate size but not the number of hardwoods. Frost et al. (1986) recommended high intensity summer fires to combat hardwood invasion. Kush et al. (2000)

concluded winter burns have similar benefits to summer burns depending on management objectives.

Fire effects studies can be problematic since changes due to fire can be difficult to detect over a short period of time and is dependent on season, frequency, and intensity of fires (Waldrop et al. 1992). Furthermore, it is difficult to replicate the effects of fire as a treatment and maintain a long term study.

From 1989 to 1993, data was collected in east Texas, including plots at Big Thicket National Preserve, to assess the effects of fire across a moisture gradient of vegetative communities. One of the objectives was to predict long-term change over the short duration of the study (Liu et al. 1997b). Although fire intensity data collected were incomplete, they concluded intensity differences caused the decrease of fire effects from dry to wet vegetative types. The impact of fire on herbaceous vegetation was not addressed in the study.

Big Thicket National Preserve started a fire effects monitoring program in 2001 and decided to continue to measure the plots from Liu et al.'s study. The purpose of this program is to facilitate adaptive management by documenting the effects of prescribed burns on the vegetation and determining if the prescribed fire program is meeting burn objectives. Long term data will better determine vegetative community changes caused by fire management, and help make recommendations on the future management of prescribed burn units. This study will examine how an altered longleaf pine ecosystem responds to managed fire disturbances in east Texas by analyzing the fire effects data of Big Thicket National Preserve.

OBJECTIVES

The objectives of this study were to:

1. Analyze fire effects data collected from the early 1990's through 2012 to demonstrate changes in forest structure and species composition from successive prescribed fire treatments.

2. Across a gradient of vegetation types, analyze long-term species abundance data to determine if prescribed fire treatments have affected the overstory tree composition.

3. Use species abundance data to determine how vegetation types have responded to varying levels of prescribed fire treatment among different size classes of vegetation over time.

LITERATURE REVIEW

Longleaf Historical Range and Fire History

The longleaf pine ecosystem is considered critically endangered, since less than three percent of these forests remain in a historical range that spanned from southern Virginia to eastern Texas (Frost 1993, Means 1996, Kush et al. 1999). Furthermore, most of the remaining longleaf communities are in an unhealthy state as a result from land development, extensive logging and the establishment of non-native pine plantations, in addition to hardwood competition and fuel accumulation from decades of fire suppression. Fire is a critical ecological disturbance required to maintain the community structure of the longleaf pine ecosystem that once dominated the southeastern United States (Waldrop et al. 1992, Glitzenstein et al. 1995, Barnett 1999).

In the early 1930’s, Chapman (1932) warned that without periodic fires, longleaf pines will decline due to competition with fire intolerant hardwood species. Chapman also stated “as long as the prevailing conditions which created this pure type continue, the longleaf pine type is as truly a climax as the beech-birch-maple type in the northern hardwoods.” In contrast, many early researchers considered the beech magnolia community as the potential climax vegetation for the forests of the southeastern coastal plain, even though the beech magnolia forests occur in a small portion of the region (Harcombe et al. 1993).

Before Native American settlement, fires caused by lightning were frequent throughout the spring and summer (Platt et al. 1991, Frost 1993). Smith (2001) estimated fire to have burned through the longleaf pine savannas on a one to five year interval, and Wahlenberg (1946) predicted an average irregular frequency of two to three years. After Native Americans arrived, fires were intentionally set in the fall, late winter, and early spring to open the forests to improve wildlife foraging, hunting, and control insect pests. This kept the fuel loads low, reducing the probability of more severe wildfires (Outcalt 2004). Once European settlers colonized the region, they continued to utilize fires for similar reasons and to improve livestock grazing. However, over a period of 250 years the longleaf pine ecosystem was drastically altered as land was cleared for agriculture, forests were logged, and fires suppressed (Johnson and Gjerstad 1998). Logging operations harvested nearly all the remaining longleaf in the West Gulf Coastal Plain by 1930 and with the lack of a seed source and feral hog pressure, regeneration was minimal (Rudolph 2000).

Dependence on Fire

Many accounts of early explorers of the southeast remarked on the expansive open canopy pine forests that allowed for a rich grassy understory obviously maintained by fire (Chapman 1932, Waldrop et al. 1992). Fire has been viewed as the most significant factor influencing longleaf regeneration (Wahlenberg 1946). Upon examining the reproductive characteristics of longleaf, Landers (1991) noted “that prepared seedbeds are common, so pressure for reproductive readiness is not great” since the trees exhibit delayed sexual maturity, have large short dispersing seeds, and infrequent/variable masting. Longleaf pine seedlings are adapted to withstand fire since the seedling remains in a “grass” stage for two to ten years. In this stage, the seedling will develop a deep taproot, a thick layer of bark, and long needles to insulate the bud from fire. The taproot helps with energy storage to replace needles damaged by fire, and to foster the bolt of growth as much as three to four feet per year, as the seedling quickly attempts to out compete ground level vegetation and avoid flame damage to the terminal bud (Chapman 1932, Wahlenberg 1946). The saplings are vulnerable to mortality by fire during the bolting period until they reach six feet in average height (Haywood 2002).

The herbaceous vegetation in the longleaf pine ecosystem can be extraordinarily diverse and is critical to longleaf pine restoration for providing the fine fuels necessary to maintain a frequent fire regime (Walker 1998). Longleaf pine forests that are open canopied support the highest plant species richness in North America (Provencher et al. 2001). Many plants native to fire-adapted ecosystems survive and reproduce following frequent fires (Drewa et al. 2002). In a wet slash pine savanna in Florida, Brewer (1998) found that species diversity decreased as the vegetation became closer to overstory trees, especially carnivorous plants, and concluded low densities of pines would facilitate the species richness of herbaceous vegetation.

Effects of Fire (Intensity, Seasonality, and Frequency)

In fire suppressed systems, species composition and community structure are driven by species distribution, available seed source, and edaphic conditions (Myers and White 1987). Haywood and Grelen (2000) found that when fire was excluded in a longleaf pine flatwood of the Kisatchie National Forest natural loblolly dominated the overstory, while hardwoods developed into a midstory. Pine litter and the herbaceous vegetation carry fire well; however, hardwood invasion due to fire suppression can lessen the intensity of subsequent fires and result in further proliferation of hardwoods (Platt et al. 1991). Drewa et al. (2002) point out that “high densities of hardwoods are likely to diminish the herbaceous species diversity”.

Glitzenstein et al. (1995) studied the effects of fire regime and habitat dynamics on Florida longleaf pine savannas in both the flatwoods and sandhill vegetative communities. They found that there were few predictable effects in relation to season or frequency (annual versus biennial) of burns, and that the variation in the communities was better related to fire intensity. The US Forest Service in east Texas found that growing season headfires killed the tops of more small hardwoods than did backfires (Hodgkins 1958). In the sandhill community, which is the driest habitat of longleaf pine, density regulated the populations (Glitzenstein et al. 1995) due to higher stocking which provides continuous needles for fuel (Platt et al. 1991); furthermore, in the flatwoods, which has poorly drained soils, competition among the pines was not significant in population dynamics. Platt et al. (1991) states that flatwoods are generally more open and the herbaceous vegetation is largely responsible for a frequent fire regime. Glitzenstein et al. (1995) documented oak mortality and top-kill was highest when burns were conducted prior to the growing season as compared to the dormant season burns, and Heywood and Grelen (2000) concluded that periodic burning later in the growing season is more effective at reducing hardwood vegetation than burns conducted early in the growing season. Additionally, it has been shown that dormant season fires have significantly increased the densities of hardwoods in longleaf pine savannas (Drewa et al.

2002). Waldrop et al. (1992) advocated annual prescribed fire to eliminate small hardwoods and develop a grassy herbaceous layer; however, it took 20 years for the change to occur. An important finding of the 43-year study that compared annual versus periodic burns in different seasons was that periodic burns minimally affected the presence of hardwood re-sprouts.

With the various effects that occur from fire intensity, seasonality, and frequency, land managers are challenged with choosing burn prescriptions that accomplish burn objectives. Furthermore, adaptive management requires that changes are monitored over time to determine the effectiveness of management and modify treatments when feasible.

Fire Helps to Resist Disturbances

Disturbance by wind events are common in the southeastern United States. Frequent hurricanes affect southeast Texas, although in some systems forest change may be chaotic and difficult to predict (Glitzenstein et al. 1986). The openness of longleaf savannas (or forests) is a direct result of storm events and fires, “which together maintain hardwood species at low densities” (Provencher et al. 2001). Landers (1991) also points out that this openness contributes to greater wind resistance.

A longleaf forest on Eglin Air Force Base, Florida was impacted by Hurricanes Erin and Opal in 1995. Burned and non-burned plots were directly and uniformly affected. Results showed that wind damage to turkey oak (*Quercus laevis*) was greater in the fire-maintained stands compared to the unburned plots, and longleaf pines suffered more wind damage on the fire suppressed plots with increased turkey oak densities. Past hurricane studies have documented similar results where the more open fire-maintained stands have the lowest damage for longleaf pines. Due to the frequent disturbances of fire, hurricanes, and the morphology of the longleaf pine,“it appears that longleaf pine is quite resilient to hurricane force winds” (Provencher et al. 2001).

Tornado occurrence density in east Texas is also very high with 5.79 tornados/10,000 km^2/year in south Hardin County, Texas, compared to a national average of 0.66/10,000 km^2/year (Glitzenstein et al. 1986). In 1983 a tornado touched down in the Hickory Creek Savanna Unit of Big Thicket National Preserve, and the vegetation response was monitored (Liu et al. 1997a). It was found that the tornado created conditions for hardwood invasion that changed the savanna community to more of a mixed pine hardwood forest. Prescribed fires that occurred after the tornado did help to reduce the amount of hardwoods; however, only half of the plots restored to savanna.

Due to the abundant amounts of resin, longleaf pine is highly resistant to southern pine beetle invasion compared to other pines. Longleaf pine is clearly the most adapted tree species for dominating a landscape frequented with fire, southern pine beetle epidemics (Rudolph 2000), and strong wind events.

Fire History of the Big Thicket Region

In the oldest known east Texas longleaf stand, tree ring analysis from 1755-1995 indicated fires occurred at frequencies of 1.5 years (Jurney et al. 2001). Recent studies indicate that variation in soils and topography are important factors in species compositions, in addition to fire history. In particular, ordination analysis confirmed vegetative communities are highly related to soil texture in the Western Gulf Coastal Plain ecoregion (Harcombe et al. 1993).

Stand history data confirm longleaf pine was an important part of the Big Thicket region in stands that were previously logged, which have succeeded into oak hickory pine stands; however, data also demonstrate oaks as a component of the forest prior to settlement (Harcombe et al. 1993).

Management Considerations/Effects of Various Management Activities

The longleaf pine ecosystem is listed as the second most threatened ecosystem reported by the United States Department of the Interior (Kush et al.

2004). Outcalt (2004) stated that recent land development has increased the wildland urban interface, which greatly complicates prescribed burn operations. This is another challenge for managers as they are strictly regulated to keep the possibility of an escaped wildfire and smoke to a minimum. Growing season fires alone may not decrease shrub densities necessary to restore herbaceous vegetation communities (Drewa et al. 2002). Managers of the longleaf pine ecosystems constantly battle the continual competition from other pines and hardwoods and need to consider different treatments, such as herbicides, in conjunction with fire (Haywood 2002). Mechanical options such as roller-chopping can also lower shrub densities to levels that growing season fires can maintain (Drewa et al. 2002). Overall, managers need to utilize the various techniques to accomplish objectives. Currently, the Big Thicket National Preserve utilizes mechanical and chemical treatments in conjunction with prescribed fire to accomplish hazardous fuels reduction with the added benefit of habitat restoration. The fire effects data will help to determine the effectiveness of prescribed burn treatments to restore longleaf pine habitat.

METHODS

Site Description

On October 11, 1974, the National Park Service established Big Thicket National Preserve as the first preserve in the National Park System. The preserve covers over 40,000 hectares in southeast Texas, and is composed of nine different land units and six water corridors. Big Thicket is located 60-120 km north of the Gulf of Mexico, between the Trinity River to the west and the Neches River to the east. The region is referred to as the West Gulf Coastal Plain, which has a subtropical humid climate with an even occurrence of precipitation throughout the year. The average annual temperature is 19.5°C with an average of 132 cm annual rainfall (Marks and Harcombe 1981). During the summer months, frequent thunderstorms and tropical storms occur. On September 24, 2005 hurricane Rita made landfall along the border between Texas and Louisiana with sustained wind speeds of 193 kph, and traveled 241 kilometers inland tracking over Big Thicket. On September 13, 2008 hurricane Ike made landfall on the northern end of Galveston Island with sustained winds of 175 kph and tracked to the northeast affecting the preserve. Various degrees of damage occurred from the hurricanes, which created gaps in the canopy and increased the amount of dead and down fuels.

This study followed methods developed by researchers from Rice University who collected data from 1989 to 1993 to determine the effects of fire management on vegetation in communities that were thought to be affected by fire. That research was conducted by Changxiang Liu as a PhD dissertation, under the direction of Dr. Paul Harcombe. After the initial research, the plots were not measured again until 2000, when Big Thicket National Preserve received funds to develop a fire effects monitoring program to support fire management. Preserve employees chose to continue measurements on the Rice University plots to allow for a long term study of fire effects. The vegetation types followed the classification of Marks and Harcombe (1981), and examined sandhill, upland pine, upperslope pine oak, midslope oak pine, lowerslope pine hardwood, and wetland pine savanna. Across each vegetation type a baseline was established, from which plot transects were randomly placed. Along these plot transects, five 10 m × 10 m subplots were established at random distances and randomly selected as to which side of the transect the subplot was located (i.e., right or left) (Figure 1). Plots were placed in both burn and control units for each vegetation type, and measured for tree, sapling, and woody seedling data.

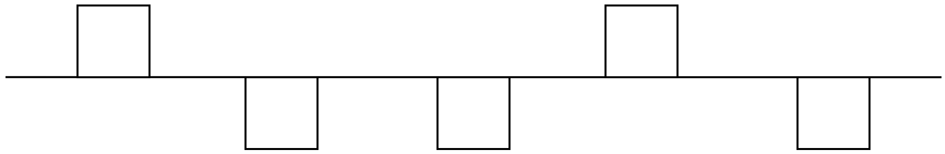


Figure 1. Plot design where each 10 x 10 meter subplot is located at random distances and random sides of a transect.

The following table (Table 1) lists the number of plots within each preserve unit for both burn and control units among the different vegetation types. Each plot includes five 10 x 10 meter subplots.

Table 1: Number of plots within each vegetation type per preserve unit. X indicates no vegetation plots were established in the preserve unit.

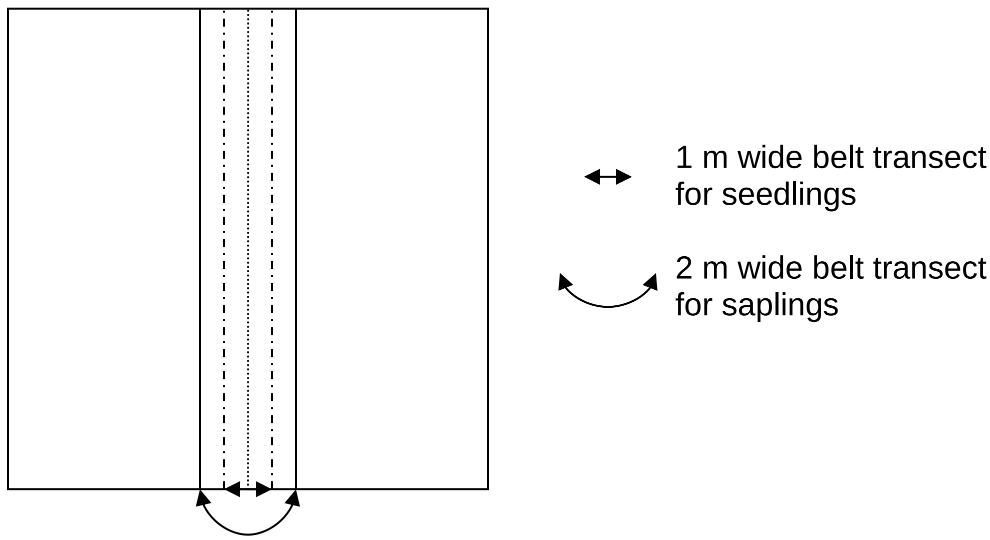
Preserve Unit	Big Sandy		Turkey Creek		Lance Rosier	
	Burn	Control	Burn	Control	Burn	Control
Upland Pine	3	1	2	1	X	X
Upperslope Pine Oak	4	4	2	2		
Midslope Oak Pine	3	3	1	1	1	1
Lowerslope Pine Hardwood	2	2	1	1	X	X
Sandhill	X	X	3	2	X	X
Wetland Pine Savanna	X	X	X	X	4	3
Total	12	10	9	7	5	4

Plot Measurements

Burn plots were measured each year following a fire, and control plots measured every 4-5 years. Overstory trees, saplings, and understory woody seedlings were measured at each 10 x 10 meter plot (Figure 2). The plot protocol followed previous methods developed by Rice University:

- Trees greater than five centimeters in diameter are classified as overstory trees and identified and measured for the diameter at breast height (dbh, 1.4 meters above soil surface). Each tree measured was tagged throughout the entire plot.
- Small trees that ranged from two to five centimeters in dbh were identified and counted but not tagged throughout the entire plot.
- Saplings that measured up to two centimeters in dbh were identified and counted over a 20 m^2 (2 m × 10 m) belt transect along the central line of each plot. Single woody stems were counted as live individuals if living tissue was present above dbh.
- Woody seedlings below 1.4 m in height were counted and identified over a 10 m^2 area (1 m × 10 m) along the central line of each plot.

Figure 2. 10 x 10 m subplot diagram demonstrating nested belt transects.



Plots were measured before a prescribed burn, immediately after a burn, one year postburn, and 2 years postburn. Since the control plots are not treated with fire, they were measured every four to five years. A plot history chart (Table 2) was compiled to record what type of measurement was conducted by plot. Reports documenting prescribed fire operations and fire effects monitoring were reviewed for each burn to reference seasonality, frequency, and intensity. The plots measured had a varying degree of prescribed fire treatment based on past management; however, the compiled reports from the history of burns helped inform conclusions when reviewing data results.

Table 2: Plot series list designating type of reading conducted for each year.

BIG THICKET FIRE EFFECTS PLOT HISTORY																	
PLOT	**FMU**	**B/C**	**1990**	**1991**	**1992**	**1993**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**
101-BSU-UP-B	1606	Burn	*i*			1	0,3,4	1	2	4	1	2	3		4	1	2
103-BSU-UPO-B	1605	Burn	*i*			1	0,3,4	1	2	4	1	2	3		4	1	2
105-BSU-LOP-B	1605	Burn	*i*			1	0,3,4	1	2	4	1	2	3		4	1	2
107-BSU-MPO-B	1610	Burn	*i*				0,3					2	3		4	1	2
109-BSU-LOP-B	1610	Burn	*i*				0,3					2	3		4	1	2
111-BSU-UP-B	1501	Burn		*i*	1	2	0		3	4	1	2		4	1	2	3
112-BSU-UP-B	1501	Burn		*i*	n	C	0	C?		4	1	2		4	1	2	3
113-BSU-UPO-B	1501	Burn		*i*	1	2	0	3		3,4	1	2		4	1	2	3
115-BSU-UPO-B	1501	Burn		*i*		1	0	3		3,4		3		4	1	2	3
117-BSU-MPO-B	1501	Burn		*i*		1	0	3		3,4		3		4	1	2	3
119-BSU-UPO-B	1201	Burn	*i*	1	2	2	0,3,4	1	2	4	1	2	4,5,6	1	2	3	
121-BSU-MPO-B	1201	Burn	*i*	1	2	2	0,3,4	1	2	4	1	2	4,5,6	1	2	3	
102-BSU-UP-C	1607	Control	*i*			C	0				C		C		C		C
104-BSU-UPO-C	1605	Control	*i*			C	0				C		C		C		C
106-BSU-LOP-C	1605	Control	*i*			C	0				C		C		C		C
108-BSU-MPO-C	1610	Control	*i*				0			C			C		C/4	1	2
110-BSU-LOP-C	1610	Control	*i*				0			C			C		C		C
114-BSU-UPO-C	1501	Control		*i*		C	0			C				C	C		C
116-BSU-UPO-C	1501	Control		*i*		C	0			C				C	C		C
118-BSU-MPO-C	1501	Control		*i*		C	0			C				C	C		C
120-BSU-UPO-C	1301	Control	*i*		C		0			C				C		C	
122-BSU-MPO-C	1301	Control	*i*		C		0			C				C		C	
301-TCU-SH-B	3601	Burn		*i*	1	2	0	3	4	1	2	4	1	2	4		2
303-TCU-SH-B	3601	Burn		*i*	1	2	0	3	4	1	2	4	1	2	4		2
305-TCU-UPO-B	3601	Burn		*i*	1	2	0	3	4	1	2	4	1	2	4		2
308-TCU-MPO-C	3601	Control		*i*		3	0	3	?	C			?	C	C		C
309-TCU-UP-B	3701	Burn	*i*	1	2	2	0					5			3		
312-TCU-UPO-B	3701	Burn	*i*	1	2	2	0			3		5			3		
313-TCU-LOP-B	3701	Burn	*i*	1	2	2	0					5			3		
315-TCU-UP-B	3101	Burn							*i*,5,6,4	1	2,3		1	2	3	4	1
316-TCU-SH-B		Burn								*i*,5,6	1	2	3	3	3		3
302-TCU-SH-C	3601	Control		*i*		C	0	C					C		C		C
304-TCU-SH-C	3601	Control		*i*		C	0		C				C		C		C
306-TCU-UPO-C	3601	Control		*i*		C	0			C			C		C		C
307-TCU-MPO-C	3601	Control		*i*		C	0	C?		C		4	C		C		C
310-TCU-UP-C	3701	Control	*i*		C		0			C			C		C		C
311-TCU-UPO-C	3702	Control	*i*		C		0			C			C		C		
314-TCU-LOP-C	3701	Control	*i*		C		0			C			C		C		C
505-LRU-WPS-B	5301	Burn	*i*	1	2	2	0,3	4	1	2	3	3	3	3	3	4	1
506-LRU-WPS-B	5401	Burn	*i*	1	2	2	0,3	4	1	2,4	1	2	3	3	3		
509-LRU-WPS-B	5401	Burn	*i*	1	2	2	0,3	4	1	2,4		3	3	3	3		
507-LRU-WPS-C	5401	Control	*i*		C		0			C			C		C		
508-LRU-WPS-C	5401	Control	*i*		C		0			C			C		C		

VEGETATION / MONITORING TYPES	
SH	Sandhill Pine Forest
UP	Upland Pine Forest
UPO	Upperslope Pine Oak Forest
MPO	Midslope Oak Pine Forest
LOP	Lowerslope Oak Hardwood Forest
WPS	Wetland Pine Savannah

B = Burn **C = Control**

PLOT WORK TYPES	
i	installation and initial readings
0	re-establishment of old plots
1	first year reading
2	second year reading
3	pre-burn reading
4	post-burn reading
5	pre-treatment (mechanical/chem)
6	post-treatment (mechanical/chem)

DATA ANALYSIS

Since the study aimed to determine changes and trends of the burned and controlled plant communities over time, data were analyzed with Canonical Correspondence Analysis (CCA), which is multivariate technique that finds the maximal separation of niches, using plots with varying combinations of species and their abundance. CCA is primarily a graphical technique (ter Braak, 1987), that projects the plots onto a set of gradients, usually chosen as the first two components that result from multivariate analysis. Subplots can then be visualized on a two-dimensional projection according to their centroids to identify locations that have similar species compositions. Liu et al. (1997b) used CCA to examine the temporal change in vegetation in relation to the prescribed burns in southeast TX, including plots established at Big Thicket National Preserve. Data for species abundance was used to compare differences in the vegetation communities. Ordination was preformed separately for each of four vegetation sizes representing different strata of the forest: trees (>5 cm DBH), small trees (2-5 cm in DBH), saplings (>0-2 cm in DBH), and seedlings (less than 1.4 m tall). Following similar methods of data analysis from the preliminary study, long term data from 1991 to 2012 was analyzed to determine how vegetation communities have responded to fire management at Big Thicket National Preserve. The analysis was performed in *R* using the vegan library (Oksanen et al., 2016).

RESULTS

Changes in vegetation over time were initially addressed in a series of ordination plots that separated the first species abundance data taken from the most recent measurement from each subplot. The graphs (Figures 3a-6d) were stratified by treatment versus control as well as vegetation size classes: overstory, small trees, saplings, and seedlings. The first analysis was followed by a second set of ordinations (Figures 7-10) that combined the earliest and latest observations that allowed assessing, via permutation tests, if there was a statistically significant difference in vegetation gradients due to time from first to last measurements, treatment versus control, and the number of burns. Each of these analyses was also stratified by the vegetation size classes of: overstory, small trees, saplings, and seedlings. The species abundance data for each of the vegetation types is encompassed in a hull identified by a dashed line. Comparing the divergence and convergence of these hulls over ordination space demonstrates how fire treatments have affected species abundance over time.

Figures 3a-3d represent the basic ordination plots for the overstory data, with separate graphs for initial and last measurements, as well as for treatment and control. The labels for the species centroids in the two-dimensional projects were omitted to allow for a clear visual of the convex hulls with labels corresponding to the vegetation types: SH for sandhill, UP for upland pine, UPO for upperslope pine oak, MPO for midslope oak pine, LOP for lowerslope pine hardwood, and WPS for wetland pine savanna. In Figure 3a, before prescribed burn treatments, most of the vegetation types had convergence of species abundance, except for the midslope vegetation type. However, in Figure 3b, only the wetland pine savanna shows distinction as a vegetation type, with more overlapping of sandhill and upland pine, as the upperslope, midslope, and lowerslope pine hardwood types converged. Figure 3c shows the first measurements taken on the control subplots and reveals more separation, especially for the wetland pine savanna and sandhill vegetation types. Figure 3d reveals that the most recent measurements for the control group and the vegetation types show increased overlap compared to the initial measurements; however sandhill and wetland pine savanna continue to be apart from the concentration of the other vegetation types.

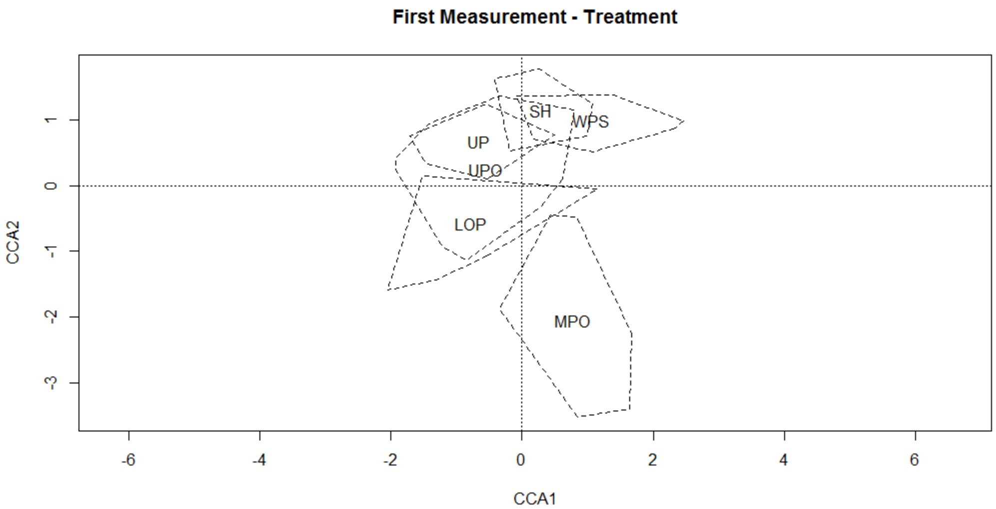


Figure 3a. Overstory – Treatment at first measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

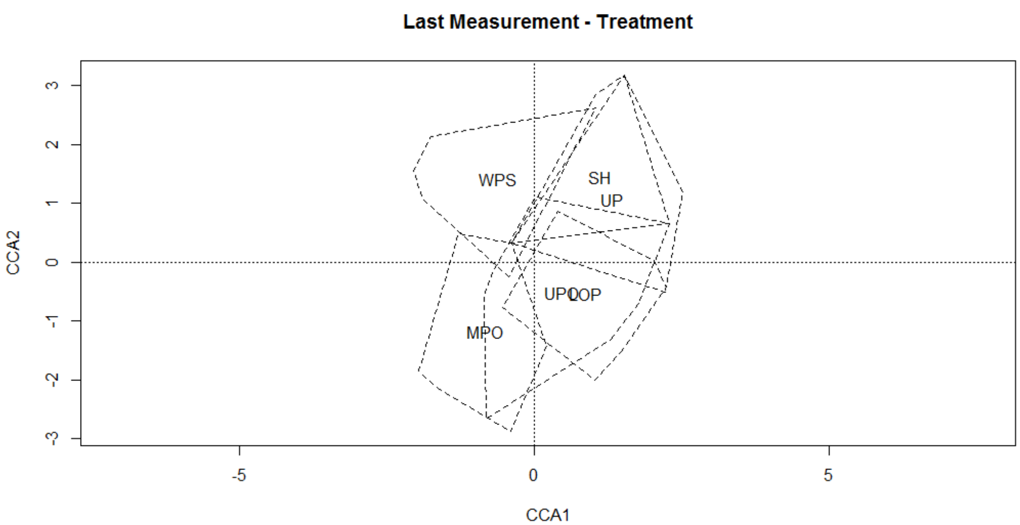


Figure 3b. Overstory – Treatment at last measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

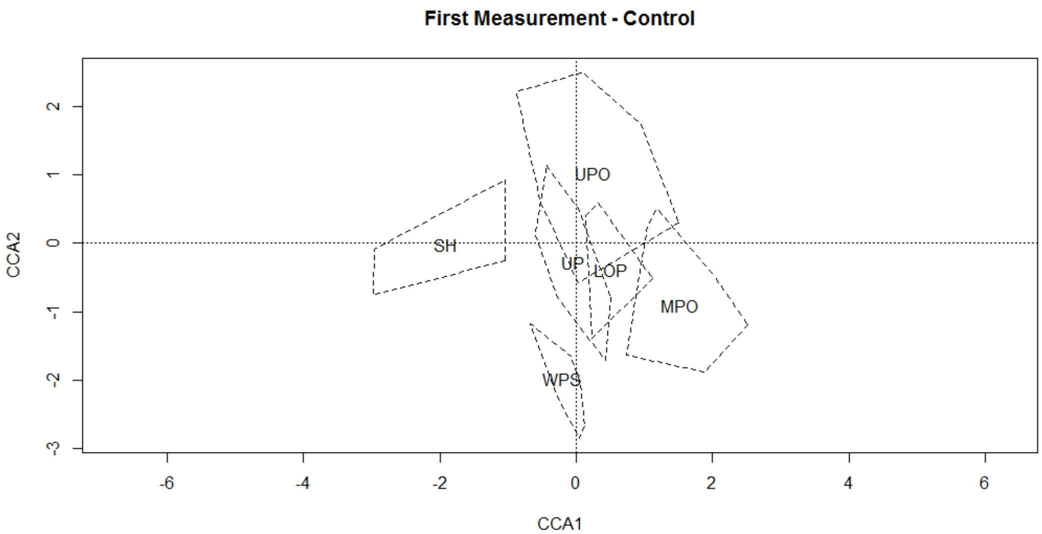


Figure 3c. Overstory – Control at first measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

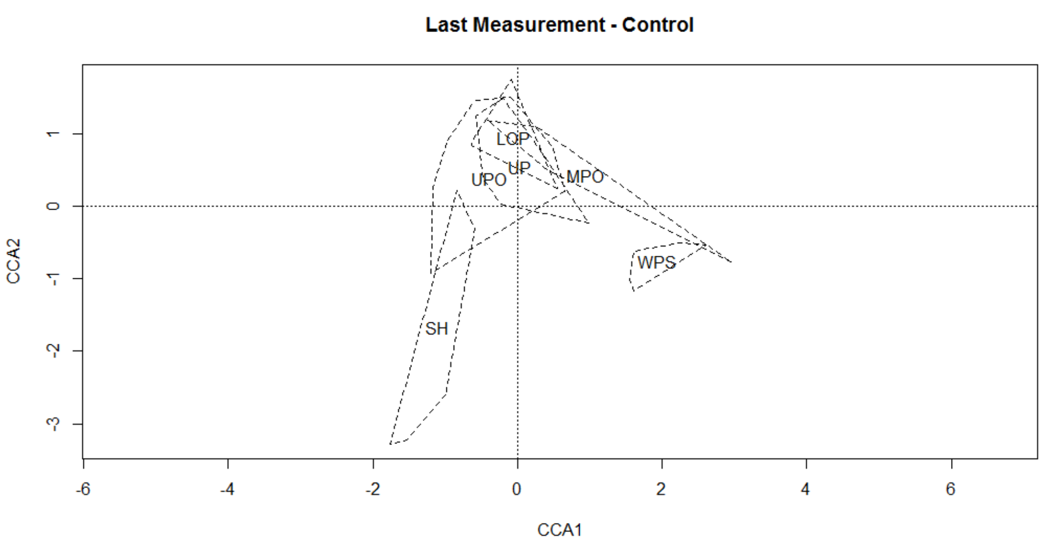


Figure 3d. Overstory – Control at last measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

Figures 4a-6d repeat the comparisons of treatment and control as well as early versus later measurements, but for the remaining three strata. Overlap between vegetation types is common, but the tendency for uniqueness of wetland pine savanna and especially sandhill appears throughout.

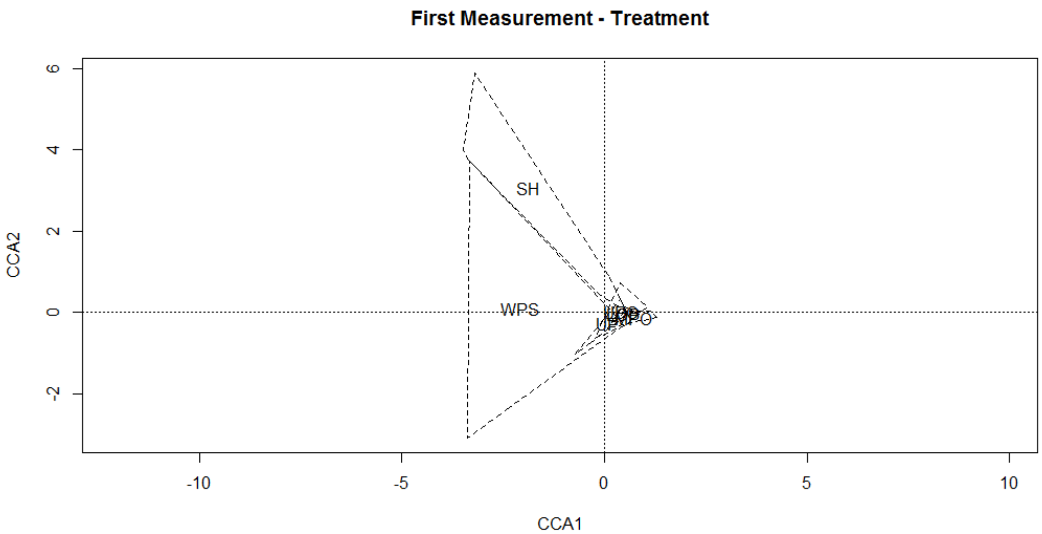


Figure 4a. Small Trees – Treatment at first measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

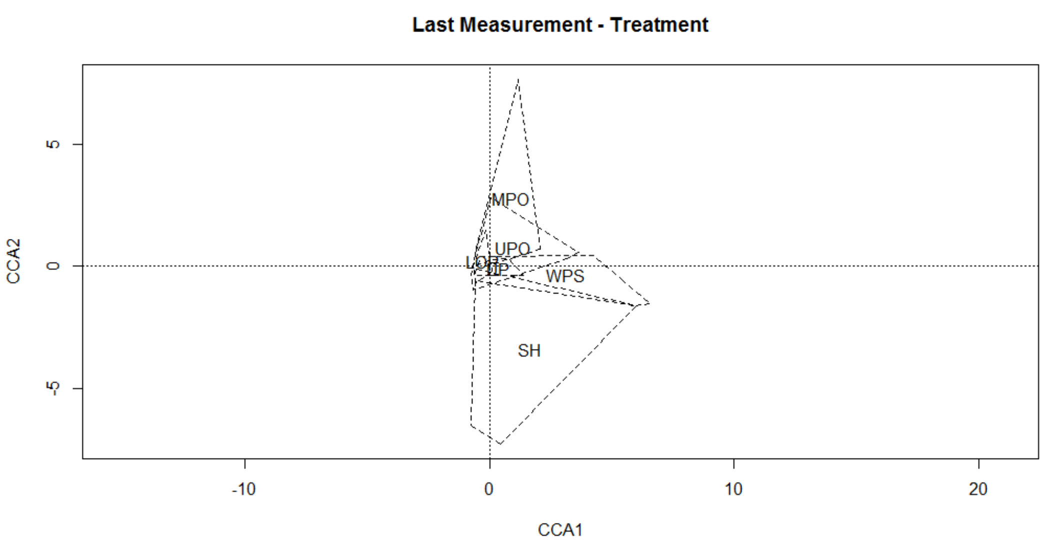


Figure 4b. Small Trees – Treatment at last measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

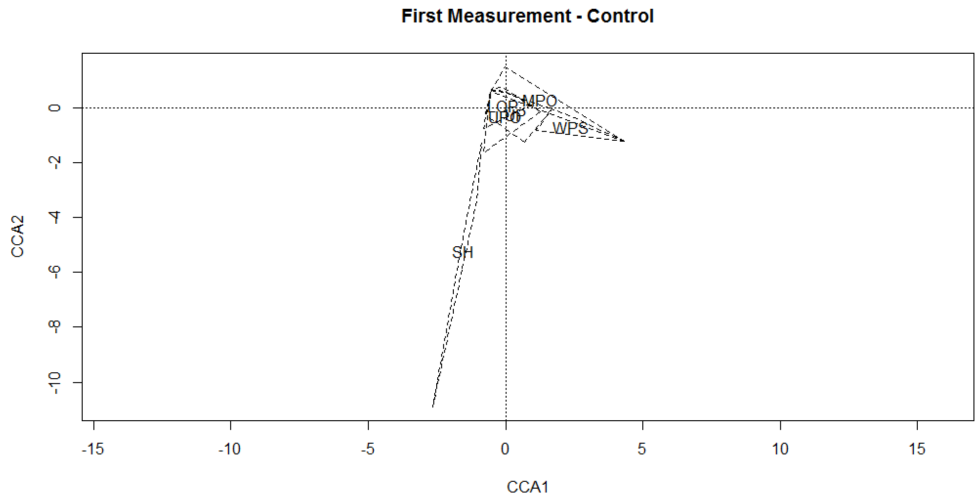


Figure 4c. Small Trees – Control at first measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

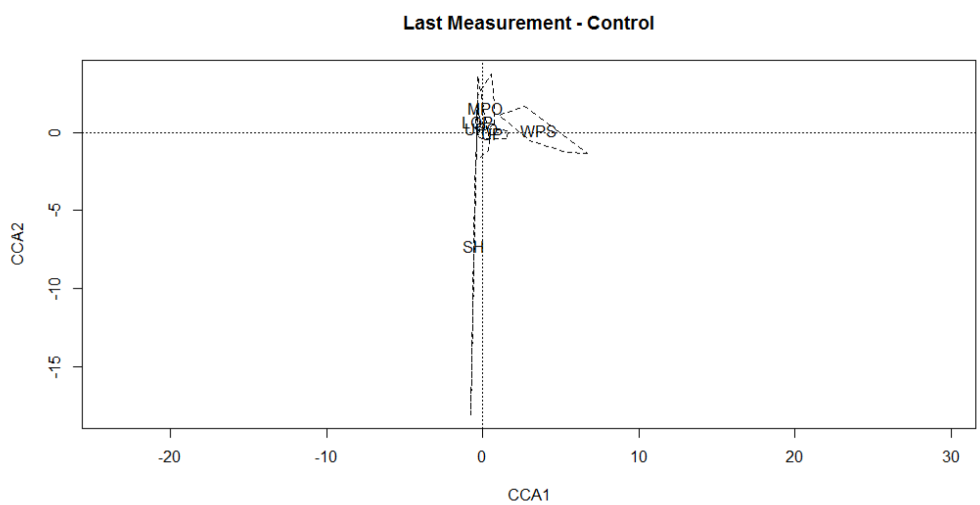


Figure 4d. Small Trees – Control at last measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

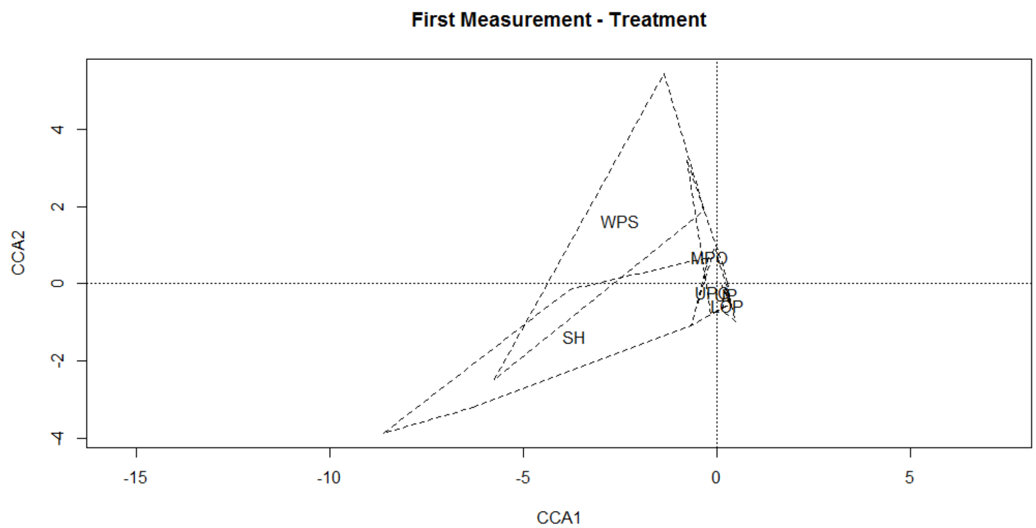


Figure 5a. Saplings – Treatment at first measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

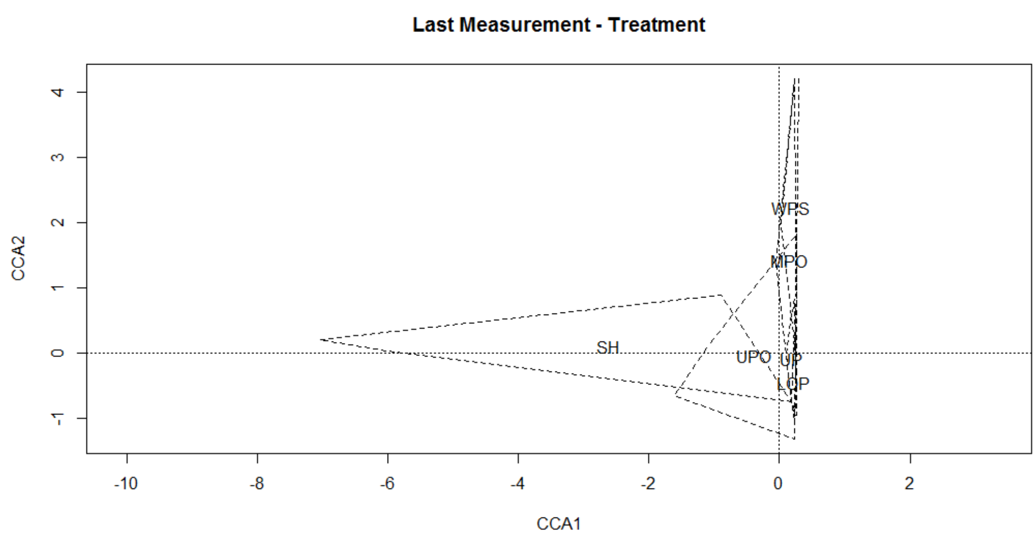


Figure 5b. Saplings – Treatment at last measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

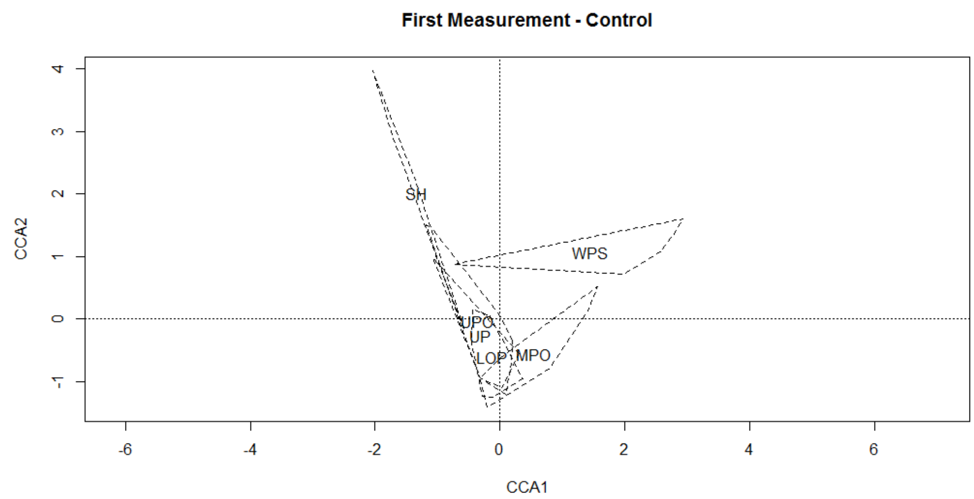


Figure 5c. Saplings – Control at first measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

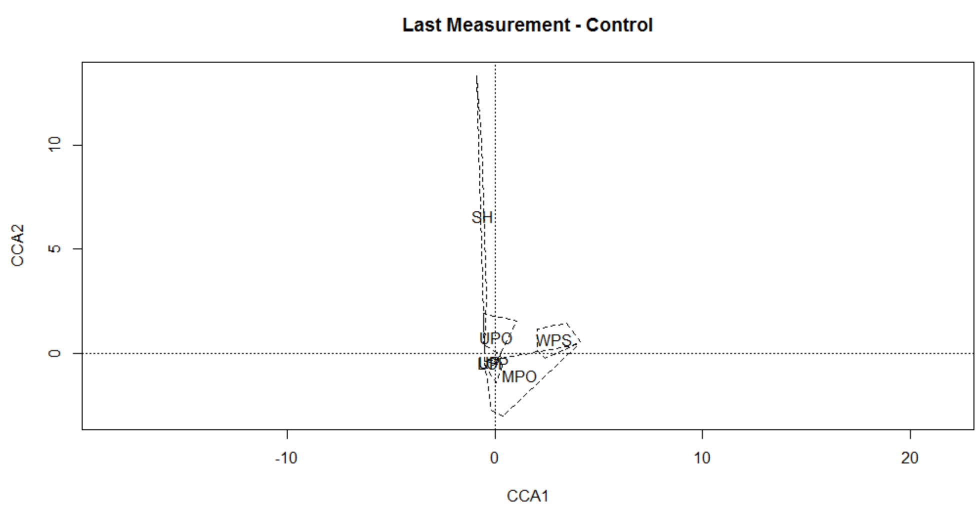


Figure 5d. Saplings – Control at last measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

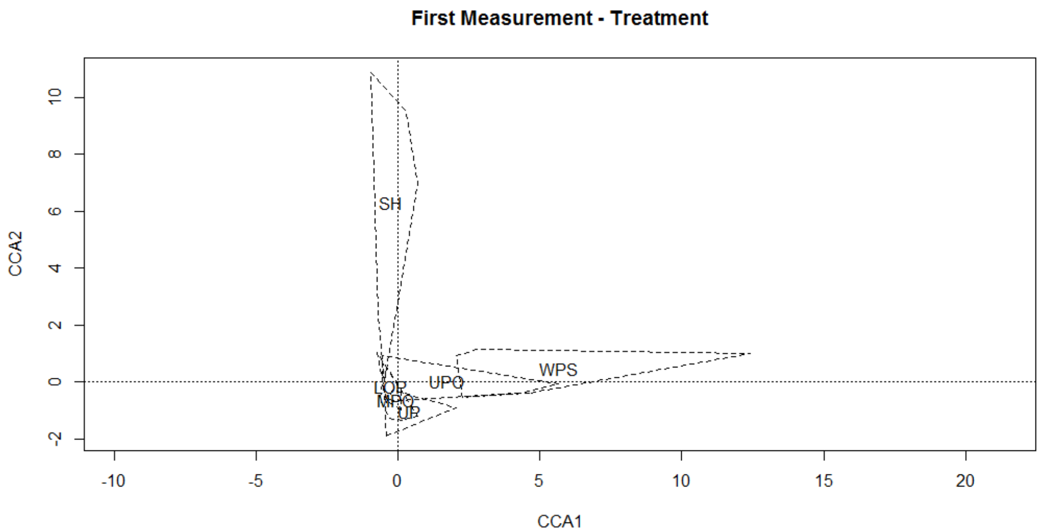


Figure 6a. Seedlings – Treatment at first measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

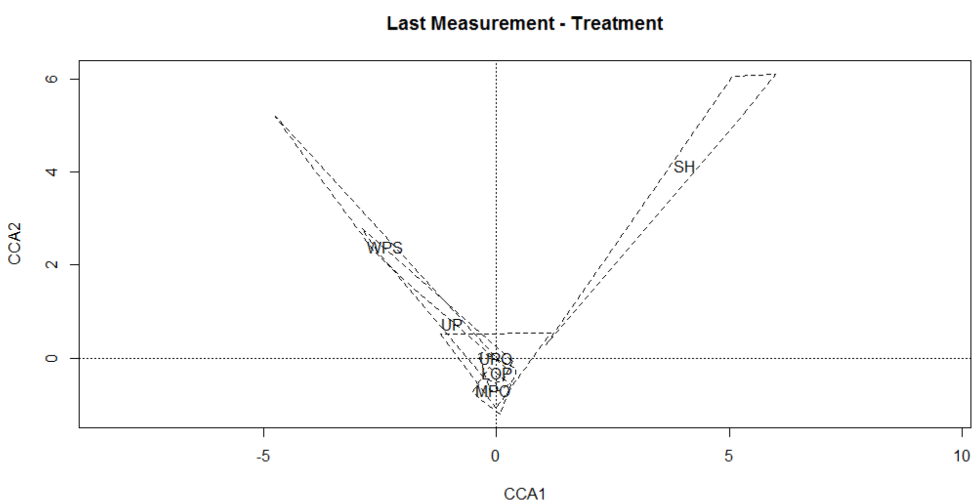


Figure 6b. Seedlings – Treatment at last measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

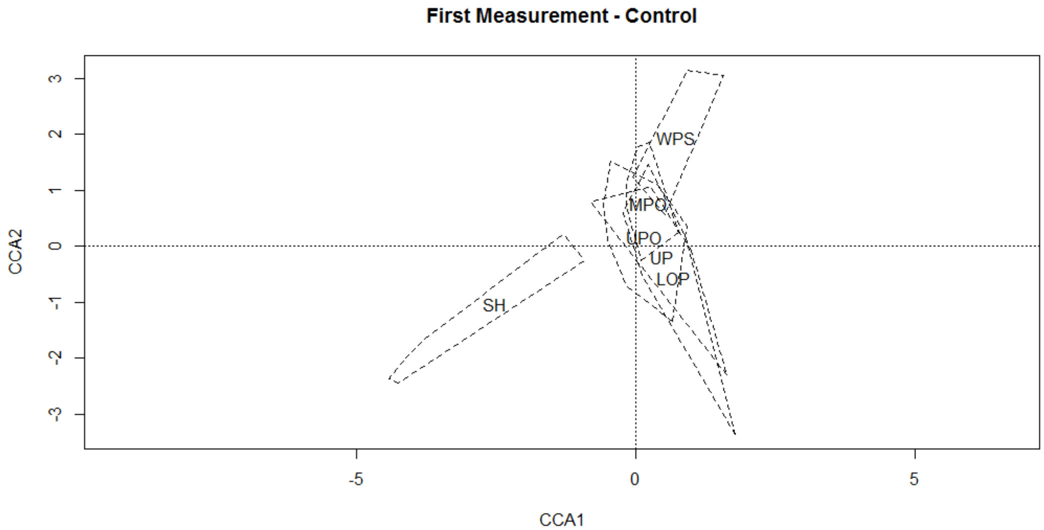


Figure 6c. Seedlings – Control at first measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

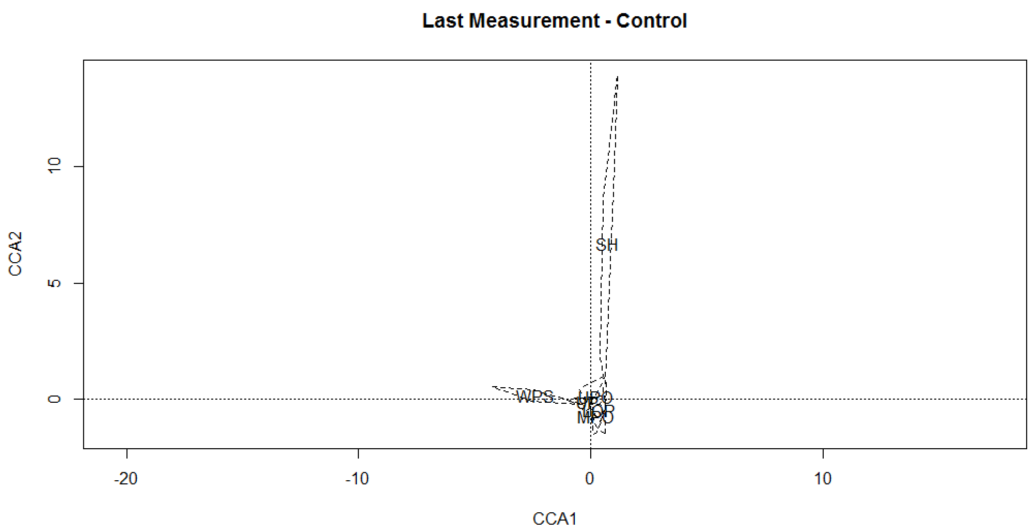


Figure 6d. Seedlings – Control at last measurement. Convex hull separates vegetation types by sandhill (SH), upland (UP), upperslope pine oak (UPO), midslope oak pine (MPO), lowerslope pine hardwood (LPO), and wetland pine savanna (WPS).

Whereas these figures provide descriptions of vegetation types along the two-dimensional gradients, they do not address the questions of treatment levels (i.e. number of prescribed burns), nor the statistical significance of time, treatment levels, and treatment versus controls. Figures 7-10 present, again stratified by plant size, subsequent CCA's that introduce these environmental variables as linear constraints. The blue convex hulls encompass the treatment versus control subplots. The red convex hulls encompass subplots that have experienced one, two, three, four, six, or seven burns (there were no cases of five burns). The labels represent the centroids of the convex hulls. The dots represent subplot observations. Table 3 shows the results of permutation tests for these variables.

Table 3: P-values from permutation tests.

	Overstory	Small Trees	Saplings	Seedlings
Number of Burns	<.001	<.001	<.001	<.001
Pre versus Post	.030	<.001	<.001	<.001
Treatment versus Control	.120	<.001	<.001	<.001

Figure 7 displays the overstory data. The treatment and control centroids separate from each other, and there is separation of the convex hulls for the different treatment levels, especially those plots with two or six burns. As Table 3 shows, there is not a significant difference between the treatment and controls. Instead, the most significant separator of subplots was the number of burns ($p < .001$). The comparison of pre versus post plot measurements was also significant ($p = .030$). The individual circles represent individual subplots all within the same vegetation type.

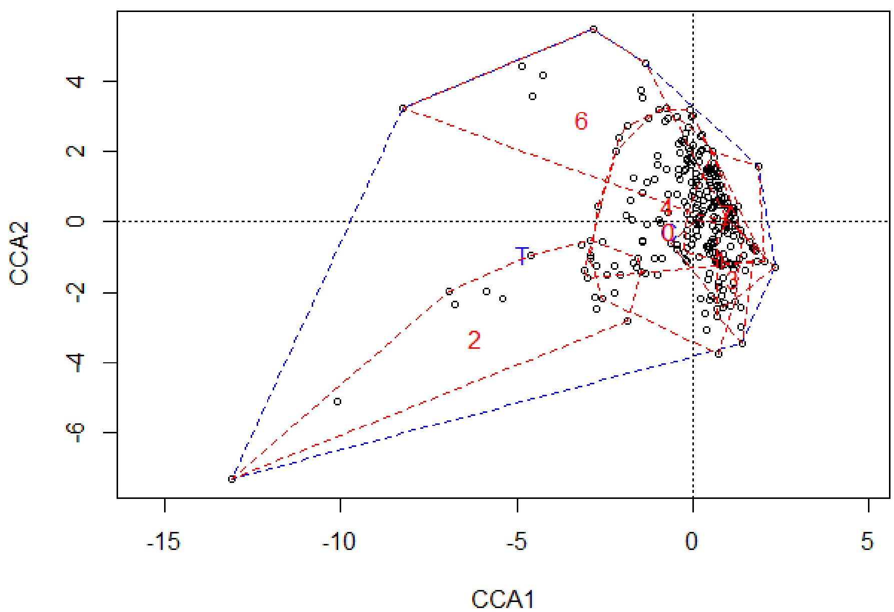


Figure 7. Overstory ordination based on pre-post, treatment assignment, and treatment levels. Blue convex hull represents treatment versus control. Red convex hull represents number of burns.

Figures 8-10 show similar plots, but for the other strata. In these cases all of the linear constraints are significant, meaning the observed differences in treatment level and burn or control all contributed to subplot separation.

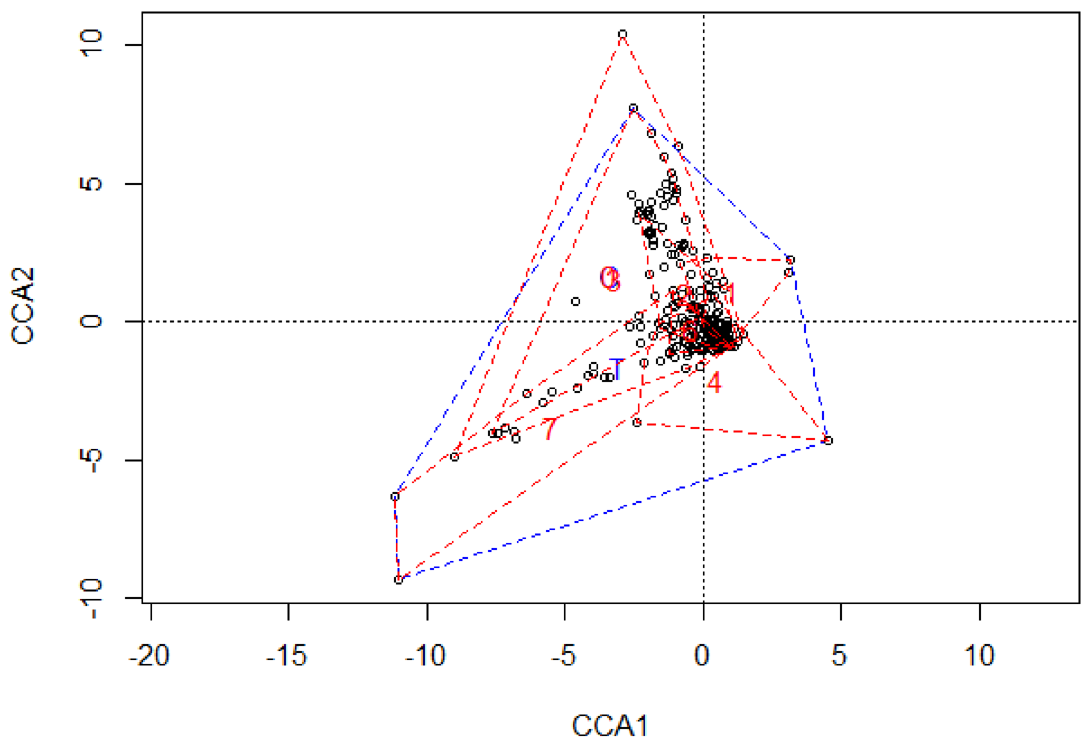


Figure 8. Small Trees ordination based on pre-post, treatment assignment, and treatment levels. Blue convex hull represents treatment versus control. Red convex hull represents number of burns.

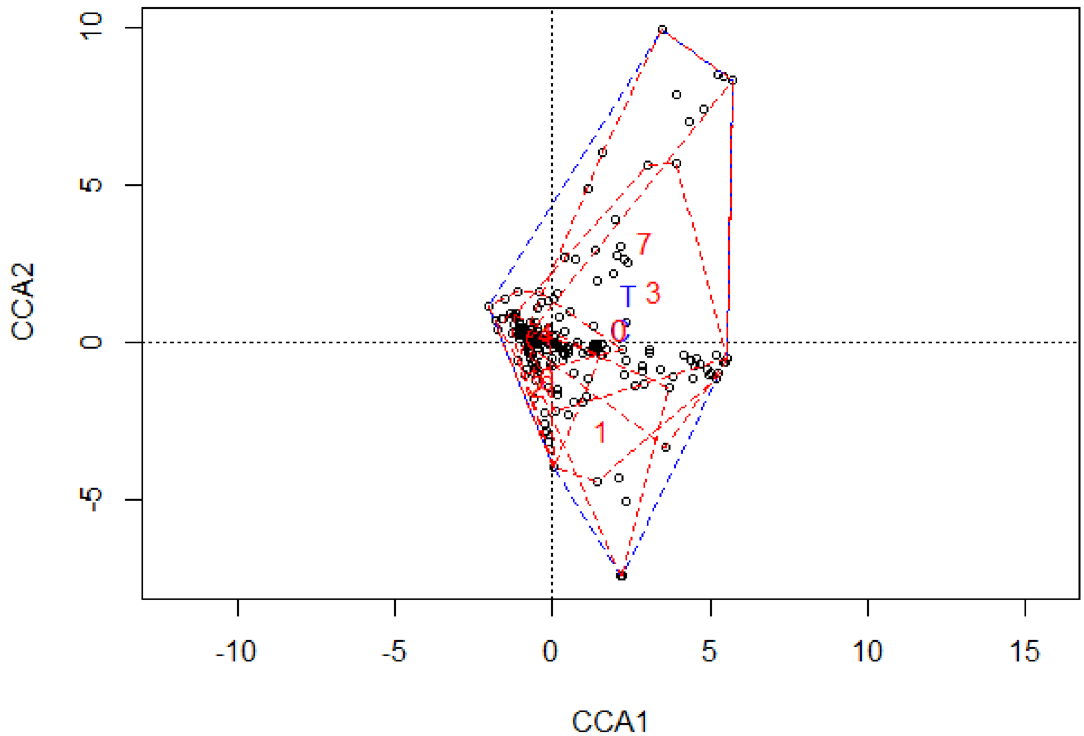


Figure 9. Saplings ordination based on pre-post, treatment assignment, and treatment levels. Blue convex hull represents treatment versus control. Red convex hull represents number of burns.

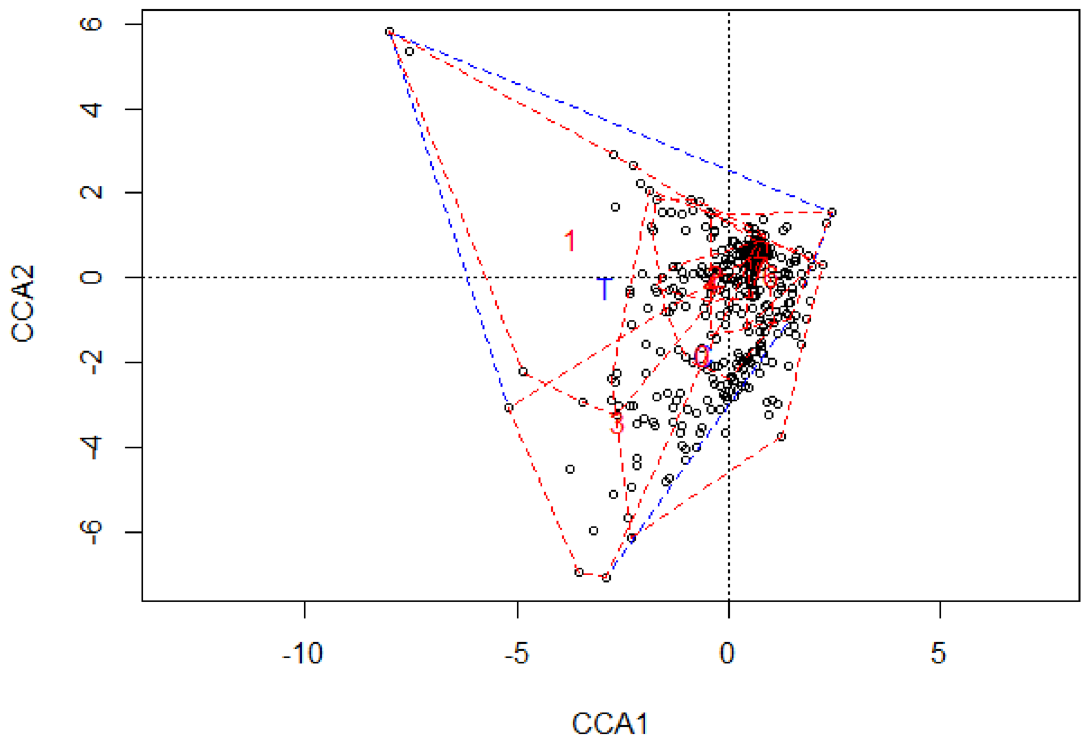


Figure 10. Seedlings ordination based on pre-post, treatment assignment, and treatment levels. Blue convex hull represents treatment versus control. Red convex hull represents number of burns.

DISCUSSION

The treatment histories of the plots were highly variable, with varying degrees of fire frequency, seasonality, and fire intensity. Also, several major natural disturbances occurred, including two major hurricanes, flooding events, and a long-term drought. Although the ordination figures and graphs appear noisy, this does represent the complicated ecological history behind the long term data collection. Changes occurred over time, the number of burns had an effect on vegetation, and that, with the exception of overstory observations, burning treatment versus controls led to differentiations over time.

Overstory

Before prescribed treatments were initiated on the burn plots, the abundance of trees did not produce distinct vegetation patterns as demonstrated by Liu et al. (1997b). Before fire management, only the midslope vegetation type and some of the wetland pine savanna types were distinct from the other community types. After two decades of fire management, the sandhill and upland pine vegetation types show more convergence and similarities in species abundance, demonstrating changes due to fire management in the more xeric communities and a trend of fire stimulated succession. The wetland savanna also showed more divergence as a distinctive vegetation type, as the overstory responded to fire management over time, even with a limited number of prescribed burns. The midslope type represents the transitional area of the forest where fire management also had an effect over time as the midslope type merged more with the slope forests demonstrating a convergence with upperslope and lowerslope forests. Over time, fire management shifted the boundaries between upperslope, midslope, and lowerslope vegetative communities, with most change in the midslope type.

The overstory control plots at the beginning of the study showed distinction between the sandhill, wetland pine savanna, and some of the midslope communities; however, the upland and slope forests converged. At the end of the study, the sandhill and wetland pine savanna continued to appear distinct; but there was some convergence with other communities, indicating encroachment by species due to an infrequent burn regime and a mosaic of fire effects despite the xeric and wet edaphic conditions that regulate vegetation communities. The midslope plots also showed less distinction over time, demonstrating successional effects and community responses to other natural disturbances not related to prescribed fire. Therefore, it is also likely that the change by the midslope communities of the burn plots is also attributed to natural succession and other environmental disturbances over time, rather than the effects of fire.

Small Trees

In Liu et al. (1997b), prescribed treatments reduced small tree densities immediately post burn, with minimal effects in the wetter vegetation types of midslope, lowerslope, and wetland savanna. Small trees data at the start of this study showed distinction among the wetland savanna and sandhill types, with the other types overlapping. At the end of the timeline, the upperslope and midslope vegetation type became more distinct. The wetland savanna type did not remain as distinct and showed more convergence with the other vegetation types, demonstrating tree encroachment, despite burning. The sandhill plots showed increased distinction as fire effected small trees, however a similar effect was not observed in the upland type. Over time, the upperslope and midslope oak pine types diverged from the other vegetation types demonstrating fire effects on a dry and mesic vegetation type.

The control plots did not demonstrate much change over time in the small trees as most types remained converged with the sandhill and wetland pine communities remaining most distinct. Although fire did not cause change in the small tree strata for the midslope and wetland savanna types in Liu's et al. (1997b) study, these results revealed change over time for these communities, demonstrating plant community release compared to the controls. Compared with smaller and larger size classes, one would predict more movement in the small trees size class. The lack of change in vegetation structure indicates that the prescribed burning was not intense enough to evoke change in the small trees. Finally, the effects of edaphic conditions in the sandhill and wetland savanna types are displayed at those communities remain distinct overtime, and when compared to overstory trees, there is less change overtime. The previous study also documented this effect among the sandhill's deep sandy soils and the dense saturated clay of the wetland savanna. These extreme soil conditions make germination more difficult and stunt plant growth, particularly for the mesic species.

Saplings

Data from the earlier study showed changes between burn and control plots; however, no clear pattern of convergence or divergence was evident, and the net changes appeared haphazard. Some of the burned stands showed species compositions returning to pre-fire conditions as species were resilient to fire and readily re-sprouted (Liu et al. 1997b). In this study, only the upperslope vegetation type appeared to demonstrate change over time with fire treatments, while the sandhill community remained distinct, retaining vegetation composition. The wetland savanna type converged with the other vegetation types, similar to the response of the small tree data, which could demonstrate encroachment over time. However, this change was not observed with the control plots as saplings data remained distinct over time. Overall, little change was exhibited in the saplings data as most of the vegetation types remained converged, with the sandhill and wetland savanna types as distinct communities over time. This demonstrates the resiliency of plant species to fire by aggressively resprouting when fire intensities do not cause mortality to saplings, and a reduced frequency of burning does not reduce species abundance over time.

Seedlings

Liu et al. (1997b) did not reveal identifiable patterns of change in the seedling strata for the vegetation types. A comparison of the longer term data reveals similar trends as most of the vegetation types treated with fire increased overlap over time, while the wetland savanna and sandhill communities remained distinct communities. The control plots demonstrated increased convergence of all the vegetation types with less distinction of the sandhill and wetland savanna types, as the plots appear to become more homogenous over time. It is harder to detect patterns over time as the seedling vegetation can be reset rapidly after a disturbance to pre-fire conditions.

Permutation Tests

When the short term data was analyzed with permutation tests by Liu et al. (1997b), no significant pattern of change was detected between burn and control or changes between vegetation types for the overstory, large sapling, and seedling data. The ordination of the small tree abundance suggested change mainly for the drier vegetation types. In comparison, the long term analysis showed significance for the changes before and after treatments among the vegetation types, and among different frequencies of burn treatments for all of the size classes. When comparing changes from burn versus control plots, all of the sizes showed significant differences except within the overstory size class. This is similar to Liu et al. (1997b), as there was not a significant change detected when the overstory tree data was compared from the burn to the control plots, indicating that fire treatments did not have an effect on the overstory vegetation after decades of treatment.

Figures 7-10 show the two-dimensional representation of the ordination data of the interactions of first to last measurements, treatment type (burn versus control), and the level of treatment (burn frequency). Figure 7 displays the overstory data, where the treatment and control centroids do separate from each other, and the convex hulls for the different treatment levels, especially those plots with two or six burns, also show separation. The distinction of the plots with six burns is attributed to not only a higher number of burns but also a mechanical treatment where brush and small trees were ground by a tracked machine with a rotating grinding head. This treatment was completed in 2007 and is the main reason the hull for plots with six treatments for the duration of the study is so distinctive and different in species composition. The plots that were burned only twice in 1992 and 2009 were composed of midslope and lowerslope forest vegetation and are distinctive due to the wetter vegetation type, but in addition, fire would have the least effect over time since it is harder to burn in these fuels under prescribed conditions and the low frequency of burns also keeps these plots different from the others. As Table 3 shows, however, there is not a significant difference between the overstory treatment and controls. Instead, the most significant separator of subplots was the number of burns ($p < .001$). The comparison of pre versus post was also significant ($p = .030$), indicating a change in species abundance data over time. Figures 8-10 show similar plots, but for the other strata. In these cases, all of the linear constraints are significant, meaning the observed differences in treatment assignment and treatment level all contribute to separating the subplots. Although these remaining graphs show distinction of the hulls in regards to differences in number of prescribed burn treatments, there is a great deal of convergence making it more challenging to see a difference among the plots based by number of treatment alone, especially since those groupings may include plots of different vegetation types.

CONCLUSIONS

In summary, changes have occurred over time, the number of burns has an effect, and – with the exception of overstory observations – burning has led to differences. The sandhill and wetland savanna types, representing the most xeric and wet vegetation types, show the most distinction over time, suggesting that edaphic conditions play the primary role in determining species abundance. However, these communities are also dependent on fire management as species encroachment still occurs, but at a reduced rate due to the extremely dry and extremely wet conditions. Having a longer term data set allowed for documenting a response in the overstory vegetation regarding the effects of burning and natural disturbances. Upland pine vegetation exhibited the most change from fire treatments, aligning with similar species abundance to sandhill vegetation. The midslope vegetation became less distinct overtime, indicating encroachment into the lowerslope and upperslope pine oak vegetation types. This effect was also seen in the control plots, demonstrating natural succession of hardwood species and increased brush densities in the absence of fire. However, the only permutation test that did not have significance was the relationship between the overstory burn versus control vegetation, indicating fire did not have a significant effect on species abundance for overstory trees.

After decades of various intensities and frequencies of prescribed fire, the data was also examined to see if there was an increase in longleaf pine seedlings due to fire management, since this is a primary objective for burn treatments. Most of the plots did not have a presence of longleaf seedlings, indicating that fire alone will not achieve these objectives with a maintenance burning regime. The maintenance burns of every 3-5 years have not affected forest structure favorable for longleaf pine, particularly under prescribed burn conditions, which do not occur during dry and windy conditions when wildfires would burn. Furthermore, when examining the prescribed burn history over the past couple of decades, many burn units were not treated on a 3-5 year rotation and had much longer intervals between burns. Also, after reviewing the burn boss maps from the early burn reports, the use of perimeter fire ignition did not effectively carry into the interior of the burn units. Some of the burn units experienced better fire coverage once interior strip firing was applied; however, hand ignition continued to create a mosaic pattern across the burn units. Furthermore breaks in vegetation from disturbance such as wind events or fire scars, also cause challenges for carrying a fire when fine fuels are absent. Finally, Big Thicket National Preserve has never used helicopter ignitions, which would help ensure fire treatment is effective, particularly in burn units 200 – 600 hectare in size and with diverse vegetation types.

Although the past effects of fire management have been limited, fire managers at the Preserve have implemented additional restoration treatments to restore forest conditions by grinding midstory brush and small trees and following up with herbicide application. In addition, areas of open canopy from past disturbances and lacking a longleaf pine seed source are replanted with longleaf pine seedlings. The combination of mechanical and chemical treatments in addition to regular fire has been effective in restoring herbaceous vegetation to the understory, which is the most diverse strata of longleaf pine habitat. Only two plots examined in this study experienced a one-time mechanical treatment, and this effect was seen in the overstory vegetation.

When compared to Liu's et al. 1997 study, which showed the role of fire was not evident in determining vegetation patterns, the longer term fire effects data had significance among the different size classes of vegetation in relation to number of burns, pre versus post fire treatment, and treatment versus control plots (with the exception of the overstory vegetation). However, change in vegetation over time is also expected due to succession and natural disturbances. Control plots helped demonstrate successional change in the absence of fire and most vegetation types, regardless of treatment, but did not become distinctive and remained converged overtime. The edaphic conditions of sandhill and wetland savanna continue to maintain species abundance on those sites. Of the remaining vegetation types, the upland pine communities responded to fire in the overstory tree size class, the upperslope pine oak vegetation responded in the saplings, while wetland savanna saw increased saplings encroachment aligning more with midslope oak pine vegetation.

When examining past prescribed burn history records and considering fire intensity and frequency, it is evident that the prescribed burning alone is not going to achieve longleaf pine restoration and has done little to effect species abundance and composition across the different vegetation types. However, efforts to use a combination of management treatments and restoration techniques to restore forest conditions including mechanical and chemical treatments will effectively open up the midstory vegetation and dense overstory. Although initially this method of restoration is expensive and labor intensive, following up these treatments with regular burning will help to restore the understory vegetation and maintain the structure of the longleaf pine forest.

LITERATURE CITED

Barnett, J.P. 1999. Longleaf pine ecosystem restoration: The role of fire. Journal of Sustainable Forestry. 9(1/2):89-96.

Brewer, J.S. 1998. Patterns of plant species richness in a wet slash-pine (*Pinus elliottii*) savanna. Journal of the Torrey Botanical Society. 125(3):216-224.

Chapman, H.H. 1932. Is the longleaf type a climax? Ecology. 13:328-334.

Drewa, P.B., W.J. Platt, and E.B. Moser. 2002. Fire effects on resprouting of shrubs in headwaters of southeastern longleaf pine savannas. Ecology. 83(3):755-767.

Frost, C.C., J. Walker, and R.K. Peet 1986. Fire-dependent savannas and prairies of the southeast: original extent, preservation status and management problems. *In*: Kulhavy, D.L. and R.N. Conner (Eds), Proceedings of a symposium: Wilderness and Natural Areas in the Eastern United States,13 May, 1985, Nacogdoches, p. 348-357.

Frost, C.C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. *In*: Hermann, S.M. (ed.) Proceedings of the 18th Tall Timbers Fire Ecology Conference, The longleaf pine ecosystem: ecology, restoration, and management. Tall Timbers Research Station, Tallahassee, Florida. pp. 17-43.

Goebel, P.C., B.J. Palik, L.K. Kirkman, M.B. Drew, L. West, and D.C. Pederson. 2001. Forest ecosystems of a Lower Gulf Coastal Plain landscape: Multifactor classification and analysis. Journal of the Torrey Botanical Society. 128(1):47-75.

Glitzenstein, J.S., P.A. Harcombe, and D.R. Streng. 1986. Disturbance, succession, and maintenance of species diversity in an east Texas forest. Ecological Monographs. 56(3):243-258.

Glitzenstein, J.S., W.J. Platt, and D.R. Streng. 1995. Effects of fire regime and habitat on tree dynamics in north Florida longleaf pine savannas. Ecological Monographs. 65(4):441-476.

Harcombe, P.A., J.S. Glitzenstein, R.G. Knox, S.L. Orzell, and E.L. Bridges. 1993. Vegetation of the longleaf pine region of the west gulf coastal plain. *In*: Hermann, S.M. (ed.), Proceedings the 18th Tall Timbers Fire Ecology Conference, The longleaf pine ecosystem: ecology, restoration, and management. Tall Timbers Research Station, Tallahassee, Florida. pp. 83-104.

Haywood, J.D. and H.E. Grelen. 2000. Twenty years of prescribed burning influence the development of direct-seeded longleaf pine on a wet pine site in Louisiana. Southern Journal of Applied Forestry. 24(2):86-92.

Haywood, J.D. 2002. Delayed prescribed burning in a seedling and sapling longleaf pine plantation in Louisiana. *In* Outcalt, K.W. (ed.), Proceedings of the eleventh biennial southern silvicultural research conference. Gen. Tech. Rep. SRS-48. Ashville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. pp.103-108.

Hermann, S.M. 2001. A brief overview of fire and season of burn in native longleaf pine ecosystems. *In* Kush, J.S. (ed.), Proceedings of the 3rd Longleaf Alliance Conference, Forest for our future – restoration and management of longleaf pine ecosystems: silvicultural, ecological, social, political, and economic challenges. Alexandria, LA. Longleaf Alliance Report No. 5. pp. 53-57.

Heyward, F. 1939. The relation of fire to stand composition of longleaf pine forests. Ecology. 20(2):287-304.

Hodgkins, E.J. 1958. Effects of fire on undergrowth vegetation in upland southern pine forests. Ecology. 39(1):36-46.

Johnson, R. and D. Gjerstad. 1998. Landscape-scale restoration of the longleaf pine ecosystem. Restoration and Management Notes. 16(1):41-45.

Jurney, D., R. Evans, J. Ippolito, and V. Bergstrom. 2001. Fire history of the West Gulf Coastal Plain. *In*: Kush, J.S. (ed.), Proceedings of the 3rd Longleaf Alliance Conference, Forest for our future – restoration and management of longleaf pine ecosystems: silvicultural, ecological, social, political, and economic challenges. Alexandria, LA. Longleaf Alliance Report No. 5. pp. 13-15.

Kush, J.S., J.M. Morgan, and R.S. Meldahl. 1999. Slow down, don't burn too fast… Got to make that old-growth last. *In*: Kush, J.S. (ed.), Proceedings of the 2nd Longleaf Alliance Conference, Longleaf pine: A forward look. Charleston, SC. Longleaf Alliance Report No. 4. pp. 109-111.

Kush, J.S., R.S. Meldahl, and W.D. Boyer. 2000. Understory plant community response to season of burn in natural longleaf pine forests. *In*: Proceedings of Tall Timbers Fire Ecology Conference. 21:32-39.

Kush, J.S., R.S. Meldahl, and W.D. Boyer. 2004. Factors affecting survival of longleaf pine seedlings. *In*: Conner, K.F. (ed.), Proceedings of the twelfth Biennial Southern Silvicultural Research Conference. Gen. Tech. Rep. SRS-71. Ashville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. pp. 314-316.

Landers, J.L. 1991. Disturbance influences on pine traits in the Southeastern United States. *In*: Hermann, S.M. (ed.), Proceedings of 17th Tall Timbers Fire Ecology Conference, High intensity fire in wildlands. Tall Timbers Research Station, Tallahassee, Florida. pp. 61-95

Landers, J.L., D.H. Van Lear, and W.D. Boyer. 1995. The longleaf pine forests of the southeast: requiem or renaissance? Journal of Forestry. 93(11):39-44.

Liu, C., J.S. Glitzenstein, P.A. Harcombe, and R.G. Knox. 1997a. Tornado and fire effects on tree species composition in a savanna in the Big Thicket National Preserve, southeast TX, USA. Forest Ecology and Management. 91:279-289.

Liu, C., P.A. Harcombe, and R.G. Knox. 1997b. Effects of prescribed fire on the composition of woody plant communities in southeastern Texas. Journal of Vegetation Science. 8:495-504.

Marks, P.L. and P.A. Harcombe. 1981. Forest vegetation of the Big Thicket, southeast Texas. Ecological Monographs. 51:287-305.

Means, B.D. 1996. Longleaf pine forest: Importance to biodiversity. *In* Kush, J.S. (ed.), Proceedings of the 1st Longleaf Alliance Conference, Longleaf pine: A regional perspective of challenges and opportunities, Mobile, AL. Longleaf Alliance Report No. 1. pp. 12-14.

Myers, R. L. and D. L. White. 1987. Landscape history and changes in sandhill vegetation in north-central and south-central Florida. Bulletin of the Torrey Botanical Club. 114:21-32.

Oksanen, J.F, G. Blanchet, R. Kindt, P. Legendre, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, M. H. Stevens and H. Wagner. 2016. vegan: Community Ecology Package. R package version 2.3-4. https://CRAN.R-project.org/package=vegan.

Outcalt, K.W. 2001. Natural stand dynamics in longleaf pine: how climate disturbances shape the community. *In* Kush, J.S. (ed.), Proceedings of the 3rd Longleaf Alliance Conference, Forest for our future – restoration and management of longleaf pine ecosystems: silvicultural, ecological, social, political, and economic challenges. Alexandria, LA. Longleaf Alliance Report No. 5. pp. 29-31.

Outcalt, K.W. 2004. Restoring structure and composition of longleaf pine ecosystems in the Gulf Coastal Plains. *In*: Proceedings of the 5th Longleaf Alliance Conference. pp 97-100

Platt, W.J., J.S. Glitzenstein, and D.R. Streng. 1991. Evaluating pyrogenicity and its effects on vegetation in longleaf pine savannas. *In*: Hermann, S.M. (ed.), Proceedings, 17th Tall Timbers Fire Ecology Conference, High intensity fire in wildlands., Tall Timbers Research Station, Tallahassee, Florida. pp. 143-161.

Provecher, L., A.R. Litt, D.R. Gordon, H.L. Rodgers, B.J. Herring, K.E.M. Galley, J.P. McAdoo, S.J. McAdoo, N.M. Gobris, and J.L. Hardesty. 2001. Restoration fire and hurricanes in longleaf pine sandhills. Ecological Restoration. 19(2):92-98

Robbins L.E. and R.L. Myers. 1992. Seasonal effects of prescribed burning in Florida: A review. Tall Timbers Research Station, Miscellaneous publication No. 8, 96 p.

Rudolph, D.C. 2000. An overview of longleaf pine ecosystems of the West Gulf Coastal Plain. *In* Kush, J.S. (ed.), Proceedings of the 3rd Longleaf Alliance Conference, Forest for our future – restoration and management of longleaf pine ecosystems: silvicultural, ecological, social, political, and economic challenges. Alexandria, LA. Longleaf Alliance Report No. 5. pp. 11-12.

Simberloff, D. 1993. Species area and fragmentation effects on old growth forest: Prospects for longleaf pine communities. *In*: Hermann, S.M. (ed.) Proceedings of the 18th Tall Timbers Fire Ecology Conference, The longleaf pine ecosystem: ecology, restoration, and management. Tall Timbers Research Station, Tallahassee, Florida. pp 1-13.

Smith, L. 2001. The history and character of longleaf pine systems in the West. *In* Kush, J.S. (ed.), Proceedings of the 3rd Longleaf Alliance Conference, Forest for our future – restoration and management of longleaf pine ecosystems: silvicultural, ecological, social, political, and economic challenges., Alexandria, LA. Longleaf Alliance Report No. 5. pp. 1-3.

Stambaugh, M.C., R.P. Guyette, and J.M. Marschall. 2011. Longleaf pine (*Pinus palustris* Mill.) fire scars reveal new details of a frequent fire regime. Journal of Vegetation Science. 22:1094-1104.

Streng, D.R. and P.A. Harcombe. 1982. Why don't east Texas savannas grow up to forest? American Midland Naturalist. 108(2):278-294.

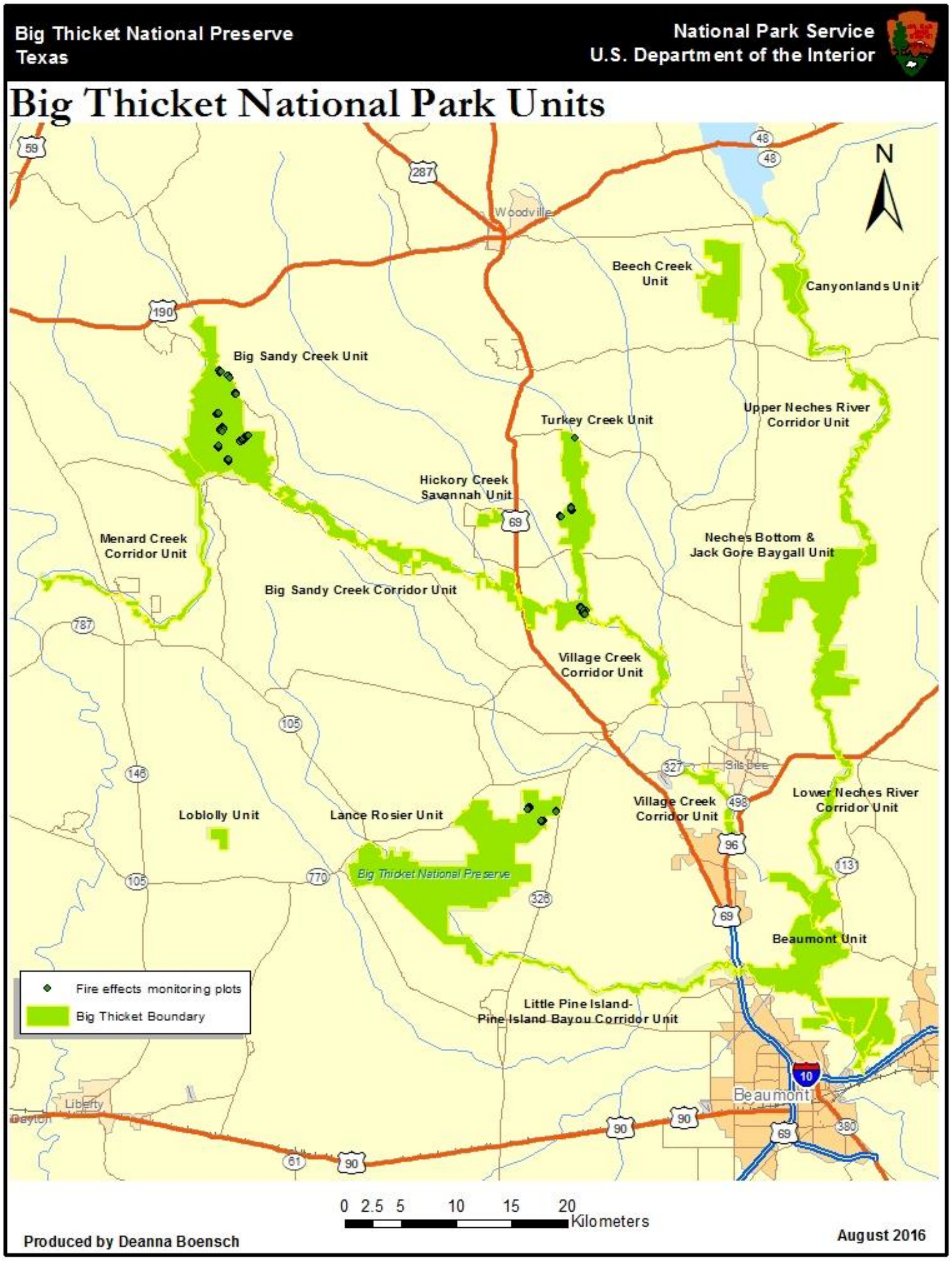
ter Braak, C.J.F. 1987. The analysis of vegetation-environment relationships by canonical correspondence anaylsis. Vegetatio. 69:69-77.

Wahlenberg, W.G. 1946. Longleaf pine, its use, ecology, regeneration, protection, growth and management. Washington, D.C: Charles Lathrop Pack Forestry Foundation in cooperation with U.S. Forest Service .

Waldrop, T.A., D.L. White, and S.M. Jones. 1992. Fire regimes for pine-grassland communities in the southeastern United States. Forest Ecology and Management. 47:195-210.

Walker, J. 1998 Ground layer vegetation in longleaf pine landscapes: An overview for restoration and management. *In* Kush J.S. (ed.), Ecological restoration and regional conservation strategies. Longleaf Alliance Report No. 3. pp. 2-13.

APPENDIX


Big Thicket National Preserve
Texas
National Park Service
U.S. Department of the Interior
Big Thicket National Park Units
Woodville
Beech Creek Unit
Canyonlands Unit
Big Sandy Creek Unit
Turkey Creek Unit
Upper Neches River Corridor Unit
Hickory Creek Savannah Unit
Menard Creek Corridor Unit
Neches Bottom & Jack Gore Baygall Unit
Big Sandy Creek Corridor Unit
Village Creek Corridor Unit
Silsbee
Village Creek Corridor Unit
Lower Neches River Corridor Unit
Loblolly Unit
Lance Rosier Unit
Big Thicket National Preserve
Beaumont Unit
Little Pine Island-Pine Island Bayou Corridor Unit
Fire effects monitoring plots
Big Thicket Boundary
Liberty
Beaumont
0 2.5 5 10 15 20
Kilometers
Produced by Deanna Boensch
August 2016

VITA

Deanna Boensch graduated with a Bachelor's degree in Biology from The College of William and Mary in 2000. The following year, she began her National Park Service career at Shenandoah National Park as a seasonal fire effects monitor and obtained her red card certification to assist with prescribed burning and wildland firefighting. In 2003, she accepted a permanent position at Big Thicket National Preserve as lead fire effects monitor and was promoted to fire ecologist in 2005. For 12 years, Deanna worked in the pineywoods of east Texas conducting fire and natural resource management duties. She witnessed ecosystems respond to restoration treatments and natural disturbances and focused much of her work on longleaf pine restoration. While working at Big Thicket, she enrolled at Stephen F. Austin State University to obtain a Master's degree and used the fire effects data she collected for her thesis.

In 2014, Deanna relocated to Natchez Trace Parkway to serve at the Natural Resource Specialist to help manage the natural resource program. Although outside of the longleaf pine range, she continues to enjoy ecosystem preservation and restoration work, especially when she can involve volunteers and future NPS stewards. She resides in Tupelo, MS with her husband, Bob, and two awesome children, Evan and Adele.