RESPONSES OF PLANT AND SMALL MAMMAL COMMUNITIES TO PRESCRIBED BURNING IN CEDAR KEY SCRUB STATE RESERVE

By

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A DISSERTATION PRESENTED TO THE GRADUATE SCHOOL
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT
OF THE REQUIREMENTS FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY

UNIVERSITY OF FLORIDA

2008

To my mother, my wife, and my three children for all their love, support, and sacrifice.
To my adviser, Dr. George Tanner, for all his help, support, and advice.

ACKNOWLEDGMENTS

The Department of Wildlife Ecology and Conservation and the Department of Environmental Protection provided the financial aid and logistic needed for carrying out the field work. Florida Fish and Wildlife Conservation Commission provided a GPS unit and participated during prescribed burning. Special acknowledgments go to my adviser, Dr. George Tanner, for his support, help, and advice when I needed it. Dr. Tanner always replies to all my requests, and he was always pleasant, cordial, and calm. I have fulfilled a very important goal in my life thanks to him. I also thank my committee members, Dr. Dick Franz, Dr. Deborah Miller, Dr. Hardin Waddle, and Dr. Alan Long for their help during the correction process.

M.S. James Collee and Jeff DiMaggio reviewed chapter drafts in advance and provided helpful comments. Dr. Waddle, Dr. Arpat Ozul, and Dr. Gary White helped me out with the MARK program. James Collee assisted me with the multivariate statistical analysis and spent time studying and analyzing the complexity of the results and their interpretations. Jeff DiMaggio and David Hoyt helped me to cut the dense scrub vegetation for making four trapping grids during three months and provided logistic help during field work. When I did not have transportation to go to Cedar Key, Jason Hall shared the field vehicle assigned to him with me and Jeff DiMaggio provided transportation in the reserve. The staff members in the department, Sam Jones, Monica Lindberg, McRae Caprice and Delores Tillman were always helpful in all my requests. Dr. Susan Carr and Zachariah Welch lent PC-ORD and books to me. All these years have been difficult, and I always had my mother, my wife, and my three children supporting and giving me their love and the incentive that I needed to complete my education.

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of the University of Florida in Partial Fulfillment of the
Requirements for the Degree of Doctor of Philosophy

RESPONSES OF PLANTS AND SMALL MAMMAL COMMUNITIES TO PRESCRIBED BURNING IN CEDAR KEY SCRUB STATE RESERVE

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May 2008

Chair: George Tanner
Major: Wildlife Ecology and Conservation

Although prescribed burning is an important management tool for ecosystem restoration in Cedar Key Scrub State Reserve, this is the first study that analyzes the effect of prescribed burning on plants and small mammals. In addition, this is the first research carried out on plant community response to prescribed fire in coastal scrub on the west side of Florida, and the 12th study about the effects of prescribed burning on small mammals in Florida. The main objectives were to determine: (a) if there were structural and compositional changes in the plant community after prescribed burning, (b) if small mammals used wetlands as temporal refugia after prescribed fire; and (c) if prescribed burning had a negative effect on the survival of the small mammal species.

The experimental design consisted of two treatment and two control sites that were sampled before and after burning from December 2003 to August 2006. Preburn vegetation samples were conducted one time in all sites, and postburn vegetation samples were carried out every three months for a 12 month period. Fifty quadrats (4 m^2 each) per site were assessed in each sampling. Resprouting was the main way of surviving and recovering from fire by the majority of the species, and almost all of the dominant species reached preburn levels during the 12 months period. This fast recovery of the vegetation after burning has been reported in the

literature but not in one year. The Detrended Correspondence Analysis showed that woody species had structural and compositional changes during the first three months postburn, but there were more compositional than structural changes after that. According to the Multi-response Permutation Procedure, the structural changes were significant; therefore, there were significant changes in absolute densities in treatment sites between pre- and 12 months postburn and between control values and 12 months postburn as a consequence of prescribed burning.

A total of 29,340 trapping nights were completed in treatment and control sites. Each site had a grid (100 traps) and a wetland next to it with two transects (10 traps each). Mice were marked to monitor movements between scrub and the vegetation surrounding wetlands during four trapping sessions before and after prescribed burning. A total of 184 individuals of *Sigmodon hispidus* (cotton rat), *Podomys floridanus* (Florida mouse), *Peromyscus gossypinus* (cotton mouse), and *Ochrotomys nuttalli* (golden mouse) were monitored during this study. In treatment sites, mice were captured mainly in the scrub (75%) before burning, they used the vegetation surrounding wetlands as temporal refugia for 11 months after burning, and they returned to the scrub after that. In control sites, mice were captured mainly in the scrub (91%) during the study. MARK analysis was only carried out on *S. hispidus* and *P. floridanus* because of the small sample size obtained for the other two species. MARK indicated that fire did not have a negative effect on the survival of *S. hispidus*. I cannot state the same for *P. floridanus* because the β parameter was not estimable. However, the data indicated that mice moved to wetlands and survived for 11 months. These results will provide guidance to managers in prescribed burning plans to establish a fire return interval according to the recuperation of the vegetation and to maintain viable populations of small mammals.

CHAPTER 1
INTRODUCTION

Fire is a natural disturbance and an important ecological factor for ecosystem management. Fire has occurred across the landscape of United States for at least 2 millions of years (Franz and Quitmyer 2005). This natural disturbance alters landscape structure, functions, and maintains biodiversity (Pyne et al. 1996). Therefore, fire is an ecological process that greatly influences composition, structure, and dynamics of many ecosystems. The ecological role of fire for ecosystem management has been appreciated because managers rely on fire history to document land management planning and silvicultural prescription, to study the effects of past fires and past fire exclusion, to simulate natural fire intervals, to perpetuate communities, and to schedule prescribed fire (Pyne et al. 1996). Particularly, prescribed fire started to be used in the southern United States before most other regions.

The prescribed burning era began after a long period of fire suppression in the southeastern United States. Fire suppression started in 1890, declined in 1930, and continued through the 1940s (Williams 2002). Prescribed burning began in the 1930s after several scientific publications supported the idea of burning wild lands. Some of these publication were "The Use and Abuse of Fire on Southern Quail Preserve," published in 1931; "Use of Prescribe Fire in Southeastern Upland Game Management," published in the Journal of Forestry in 1935; and "Relation of Burning to Timber and Wildlife" in the North American Wildlife Conference in 1935 (Kennard 2007). These publications created the political atmosphere to re-introduce prescribed burning. However, prescribed burning was discontinued in Okefenokee Swamp in 1930 (reintroduced in 1970) and in Welaka Reserve in 1935 (reintroduced in 1980). The first official prescribed burning carried out on federal land took place in Osceola National Forest in 1943 (Stanturf et al. 2002). Therefore, the process of re-introducing prescribed burning into the southeastern United States was not a one-time event. But, it started to be more frequently used in 1945 and extensively used after 1980.

During the 1980s, an estimated 16 million ha of forest land and 1.6 million ha of range and agricultural lands were treated with prescribed fire each year in the southern United States (Wade and Lunsford 1989). The majority of this treated area was for wildfire hazard reduction, wildlife habitat improvement, and range management. Nearly 0.81 million ha of rangeland were burned annually in Florida alone (Brown and Smith 2000). The main reason for carrying out such extensive prescribed burning in the Southeast, specifically in Florida, was because several natural ecosystems depend on fire. One of these ecosystems is the Florida scrub.

Florida scrub is a distinctive and threatened ecosystem (Myers 1990, Whelan 1995, Menges 1999, Brown and Smith 2000, Schmalzer 2003). It is distinct because it supports a high number of threatened and endangered plants and animals (Myers 1990, Stout and Marion 1993, Stout 2001). It is threatened because of natural fragmentation, human perturbations, and fire exclusion (Myers 1990). In addition, natural fire no longer occurs with the same intensity and frequency because the scrub habitat has been fragmented and reduced. Conservation of this unique ecosystem relies on management and research.

Management and research of the scrub habitat is essential for the survival of many plant and animal species. Prescribed burning has been the primary management technique for maintaining scrub communities because it is a fire-maintained system (Myers 1990, Whelan 1995, Menges 1999). Understanding the structural and compositional changes of the scrub communities after prescribed fire is important for making management decisions to maintain appropriate conditions for plants and animals relying on these communities (Schmalzer and Hinkle 1992a). In addition, understanding prescribed burning effects on wildlife is critical in order to provide a more comprehensive management based on the knowledge of plant and animal responses to prescribed burning. Even though the combination of management and research is needed for conservation and restoration purposes, research about plant and animal responses to prescribed fire is strongly needed in several public lands in Florida. One of these public lands is Cedar Key Scrub State Reserve (CKSSR).

Prescribed burning is the main management tool in CKSSR, and it has been intensively used since 1985. Prescribed fire is considered the most potent and critical natural resource management tool at the reserve (DEP 1998). The main objectives of the program are to restore the natural fire regimen within the reserve, create a mosaic of different successional stages, and maximize ecological diversity (DEP 1998). To achieve these objectives, CKSSR was divided into burn zones, and burn programs were assigned to each zone. In addition, these objectives were established because natural communities, and the associated plant and animal communities adapted to them, have been negatively impacted by extended periods of fire suppression. For this reason, the program targets restoring the habitat for the endangered *Aphelocoma coerulescens* (Florida scrub jay) and other species of interest such as *Podomys floridanus* (Florida mouse) and *Gopherus polyphemus* (gopher tortoise). These species have specific habitat requirements that are not satisfied by long-unburned scrubs. Proper management of the scrub habitat in the reserve will be critical for the long-term survival of these species of interest. Although prescribed burning has an important role in the reserve, no study has evaluated the effect of prescribed burning on plants or wildlife within this scrub community. Research on this topic is essential for proper management. This study is the first research conducted with the purpose of evaluating plants and small mammal responses to prescribed burning in scrubby flatwoods in CKSSR.

Study Area. CKSSR is located in Levy County, Florida, approximately 4 km east of the town of Cedar Key (Figure 1-1). It consists of 1973 ha, and it was acquired in 1978 under the Environmentally Endangered Land program (DEP 1998). Cedar Key has a warm and humid climate. Based on 30 years of weather records (Weather.com), annual temperature and precipitation average 20.8 °C and 126.3 mm (Figure 1-2), respectively, but year-to-year variability is high. For instance, based on March 2004 – February 2005 data from Accuweather.com and March 2005 – August 2006 data from a local station in CKSSR, annual temperature and precipitation average 21.0 °C and 110.4 mm, respectively (Figure 1-3). The heaviest rainfall typically takes place from June to September with some precipitation in all months of the year. Thunderstorms are frequent during the summer, and lightning strikes are common.

Hurricanes Charley (9-14 August), Frances (25 August – 8 September), and Jeanne (13-28 September) hit the Florida peninsula in 2004. Charley did not hit Cedar Key directly, but its winds brought some rainfall to the area. Frances and Jeanne hit Cedar Key directly and brought a high precipitation into the area. A total of 372.5 mm fell in Cedar Key during September (Figure 1-3). This is 204.5 mm over the monthly average precipitation (Figure 1-2). This amount of precipitation caused water from nearby wetlands to inundate the scrubby flatwoods and the sand pine scrub, and they remained partially flooded for several weeks.

Different types of soils support different plant communities. CKSSR has ancient dunes of aeolian origin (White 1970). Sand deposits in the reserve are considered to be part of the Silver Bluff Terrace (DEP 1998). The reserve has eight types of soils that range from well-drained sandy soils in the upland to poorly drained, frequently flooded, and mucky soils in tidal marsh (Slabaugh et al. 1996). Numerous wetlands are integrated with other communities in the landscape. CKSSR is a mosaic of wetlands (basin swamps, basin marshes, depression marsh, tidal marshes, hydric hammock, and estuarine), mesic flatwoods, scrubby flatwoods, sand pine scrub, and sandhill. This mosaic of habitats makes prescribed burning difficult because each habitat has a different set of optimal burning conditions. Particularly, scrubby flatwoods are surrounded by mesic flatwoods and wetlands and may occassionaly be adjacent to sand pine scrubs. Occasionally, the prescription for an upland vegetative community includes surrounding wetlands. This is one aspect of interest land managers because the vegetation surrounding wetlands might play a role as refugia after prescribed fire in the scrubby flatwoods. Scrubby flatwoods occur on sites of well-drained sandy and acid soils and low in nutrients. They represent a great percentage of the total land area in the reserve, and several of them are over-mature because the last wildfire in the area occurred in 1955 (DEP 1998).

Scrubby flatwoods are represented by several oaks, ericaceous, palm, and herb species. The most common oaks are *Quercus myrtifolia* (myrtle oak), *Q. geminata* (sand live oak) and *Q. chapmanii* (Chapman oak) (Figure 1-4). Ericaceous species are *Lyonia ferruginea* (rusty lyonia), *L. lucida* (fetterbush), and *L. fruticosa* (staggerbush) (Figure 1-5). Palms are *Serenoa repens* (saw-palmetto) and *Sabal palmetto* (sabal palm) (Figure 1-6). Herb species richness is low because of the overgrown condition of the scrubby flatwoods. Some of them are the following: *Galactia elliottii* (Elliot's milkpea), *G. mollis* (soft milkpea), *Solidago odora* (Chapman's golden rod), *Crotalaria rotundifolia* (rabbit bells), *Woodwardia virginica* (Virginia chain fern) and several *Panicum* spp (Figure 1-7). Oaks and ericaceous species dominate the scrubby flatwoods, and this is the habitat for several animal species of interest.

CKSSR has species of concern (*G. polyphemus* and *P. floridanus*) and threatened species (*Drymarchon corais couperi* (eastern indigo snake) and *A. coerulescens*) according to Florida

Fish and Wildlife Conservation Commission. This study deals with *P. floridanus* and other small mammals found in scrubby flatwoods such as *Ochrotomys nuttalli* (golden mouse), *Peromyscus gossypinus* (cotton mouse), and *Sigmodon hispidus* (cotton rat) (Figure 1-8 and 1-9). Even though CKSSR has the potential to be a study area for scientific research, it has not received the attention that it deserves from the scientific community.

Little research has been conducted in CKSSR. Amoroso (1993) carried out a floristic study, in which several carnivorous orchids species were reported. Morgan (1998) studied the association of *P. floridanus* with *G. polyphemus*'s burrows and vegetation characteristics. Several surveys for *P. floridanus* and *A. coerulescens* have been carried out. Dr. James Layne monitored the population of *P. floridanus* in one stand from 1957 to 1995. Later, Florida Fish and Wildlife Conservation Commission and the Department of Park and Recreation continued trapping *P. floridanus* in the reserve in several locations from 1995 to 1997. *A. coerulescens* has been surveyed annually since 1980. Since no study has evaluated the effects of prescribed burning on plants and small mammals, an experimental design was planned to carry out this research.

Four sites were selected to study the effects of prescribed burning. The experimental design considered two treatment sites (5C and 2M) and two control sites (5A and 5D) not selected at random (Figure 1-10 through 1-12). The park manager already had planned to burn long-unburned scrubby flatwoods in the reserve, and we visited them to do the selection. I chose four sites with the same characteristics regarding fire history, plant species composition, a wetland next to them, and no mechanical treatment (cutting or roller chopping). The other sites fulfilled the first three criteria, but they received partial mechanical treatment. The experimental design included vegetation sampling and trapping in these sites and in the vegetation surrounding wetlands next to trapping grids before and after prescribed burning. The vegetation surrounding wetlands was an ectone between the scrubby flatwoods and the proper vegetation of wetlands. This dissertation was interested in determining the potential role of the vegetation surrounding wetlands as refugia for small mammals when adjacent scrubby flatwoods are prescribed burned.

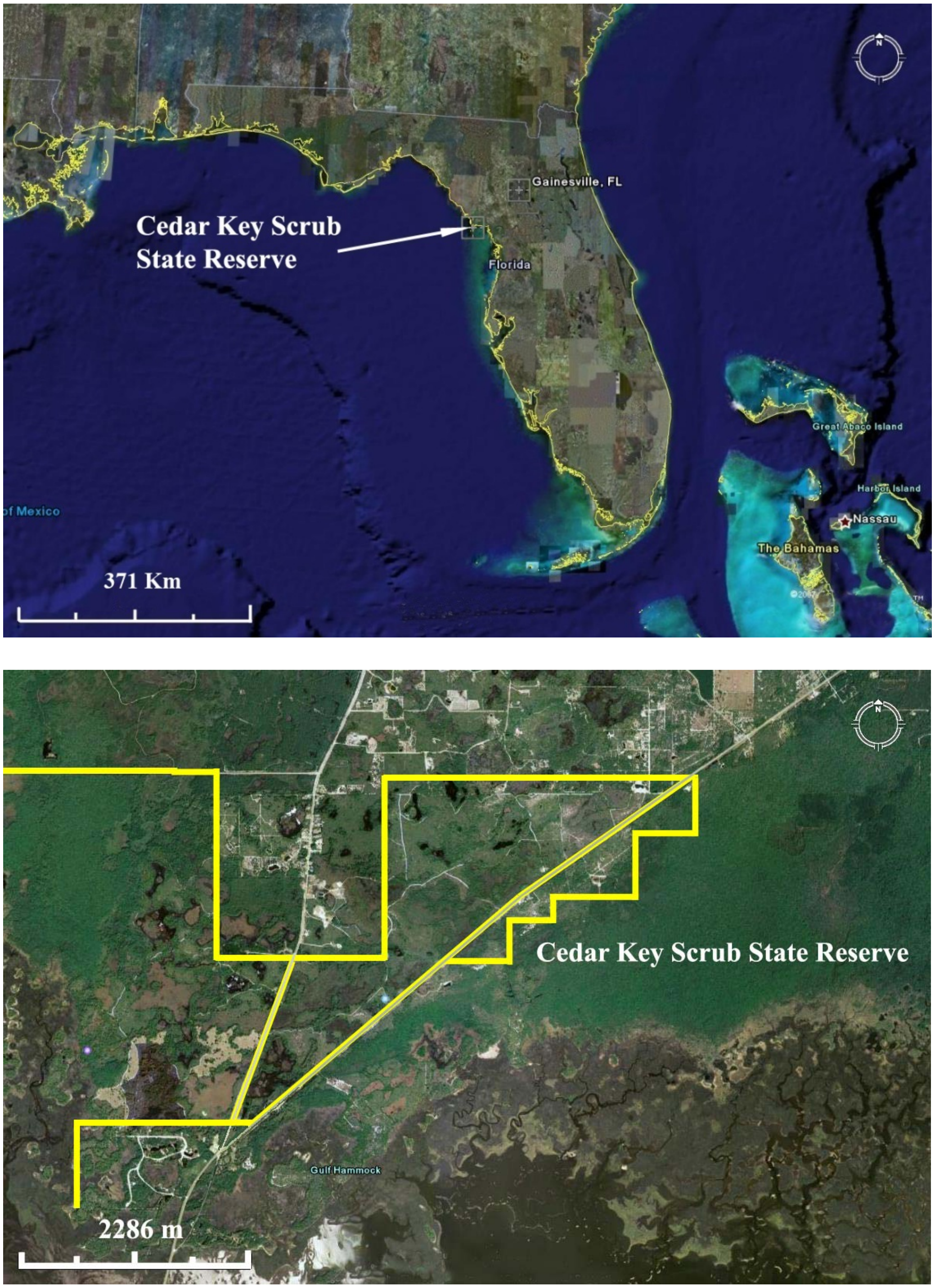


Figure 1-1. Cedar Key Scrub State Reserve. A) Location in Florida. B) Boundary of the reserve.

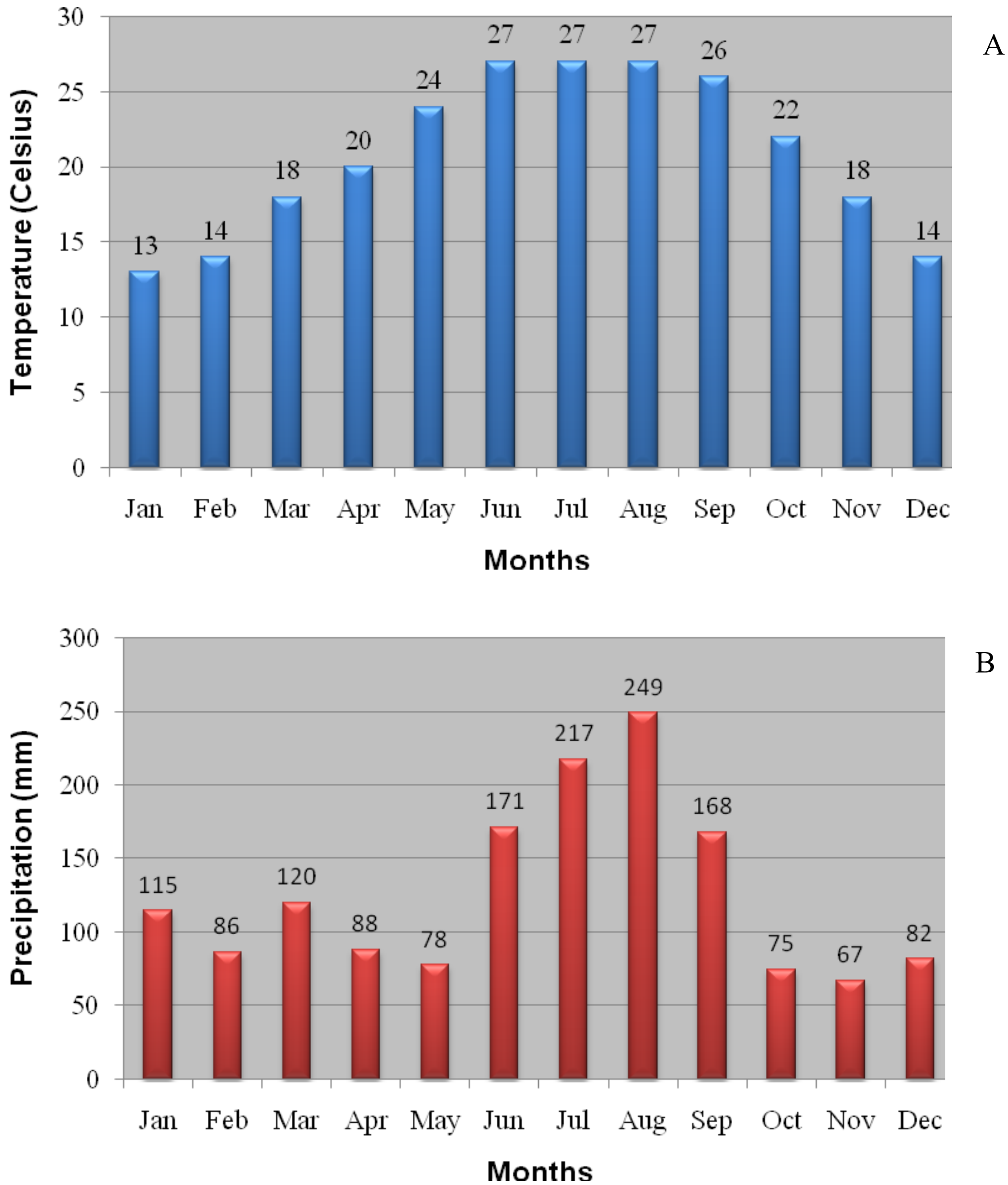


Figure 1-2. Thirty years of weather data for Cedar Key (Accuweather.com). A) Average monthly temperature. B) Average monthly precipitation.

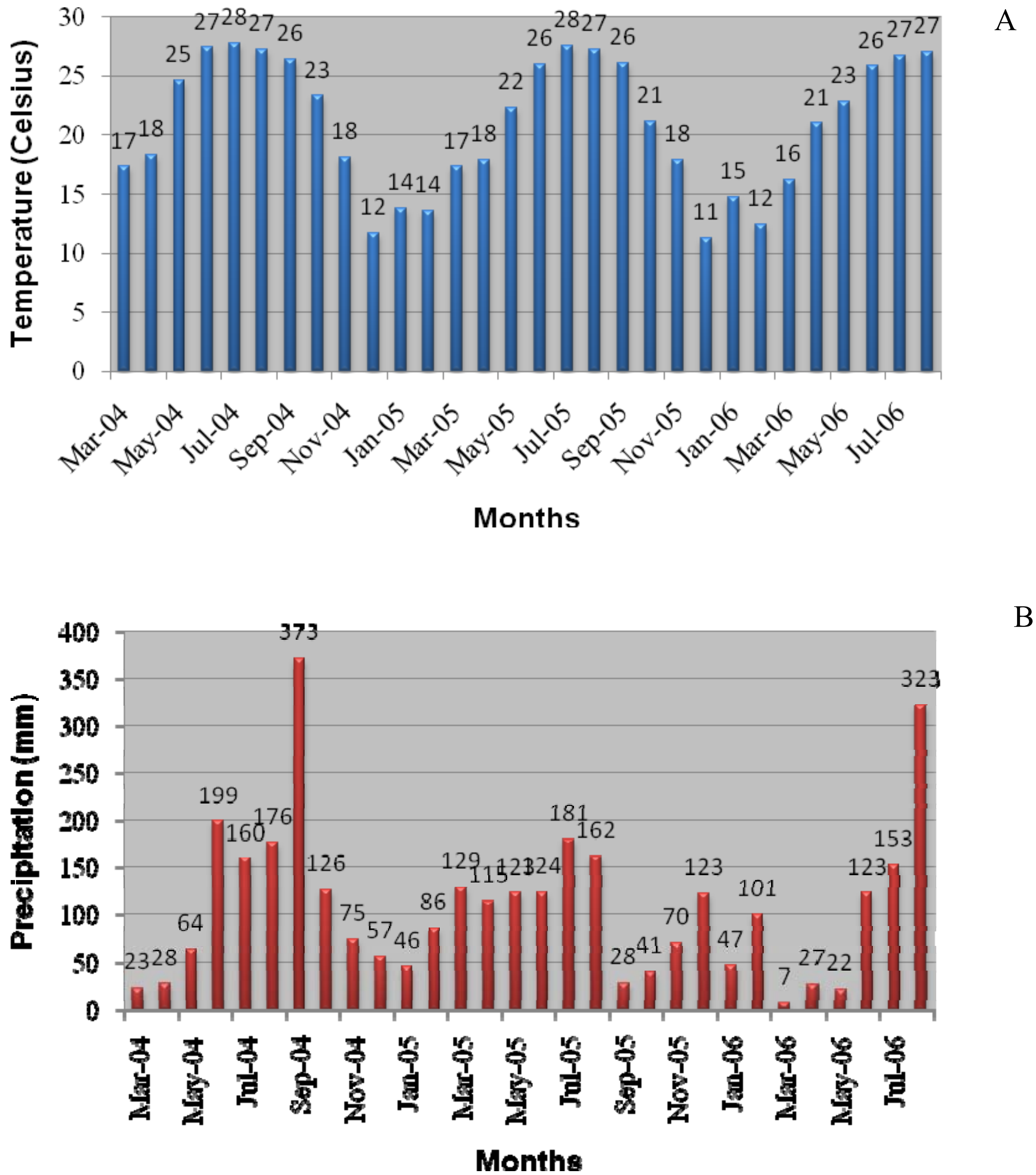


Figure 1-3. Weather data from local station in Cedar Key Scrub State Reserve. A) Monthly average temperature. B) Average monthly precipitation.

A

B

C

Figure 1-4. Three species of oaks in Cedar Key Scrub State Reserve. A) *Quercus myrtifolia* (myrtle oak). B) *Quercus geminata* (sand live oak). C) *Quercus chapmanii* (Chapman's oak).

A

B

C

Figure 1-5. Ericaceous shrubs in Cedar Key Scrub State Reserve. A) *Lyonia ferruginea* (Rusty lyonia). B) *Lyonia lucida* (fetterbush). C) *Lyonia fruticosa* (staggerbush).

A

B

Figure 1-6. Palm species in Cedar Key Scrub State Reserve. A) *Serenoa repens* (scrub palmetto). B) *Sabal palmetto* (sabal palm).

A

B

C

Figure 1-7. Herbaceous species in Cedar Key Scrub State Reserve. A) *Galactia elliottii* (Elliot's milkpea). B) *Solidago odora* (Chapman's golden rod). C) *Galactia mollis* (soft milkpea).

A

A

B

Figure 1-8. Two rodent species found in Cedar Key Scrub State Reserve. A) *Podomys floridanus* (Florida mouse). B) *Ochrotomys nuttalli* (golden mouse).

A

B

Figure 1-9. Two cotton rodents found in Cedar Key Scrub State Reserve. A) *Peromyscus gossypinus* (Cotton mouse). B) *Sigmodon hispidus* (cotton rat).

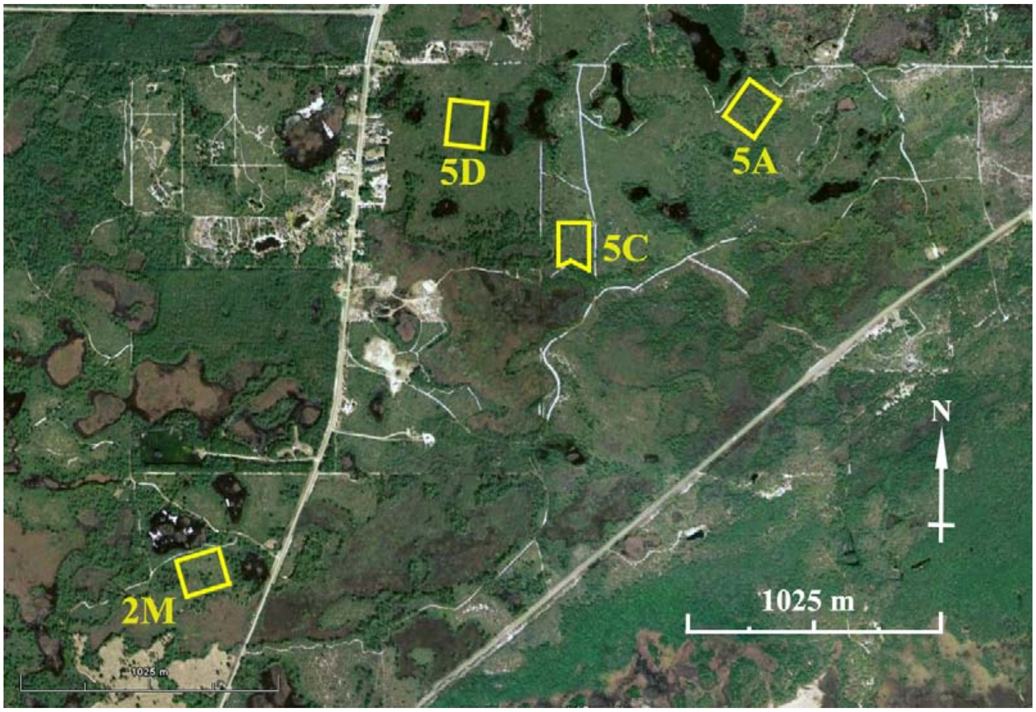


Figure 1-10. Location of treatment (5C and 2M) and control sites (5A and 5D) in Cedar Key Scrub State Reserve. Yellow blocks are trapping grids installed in each site. The actual size of each site is bigger than the grid.

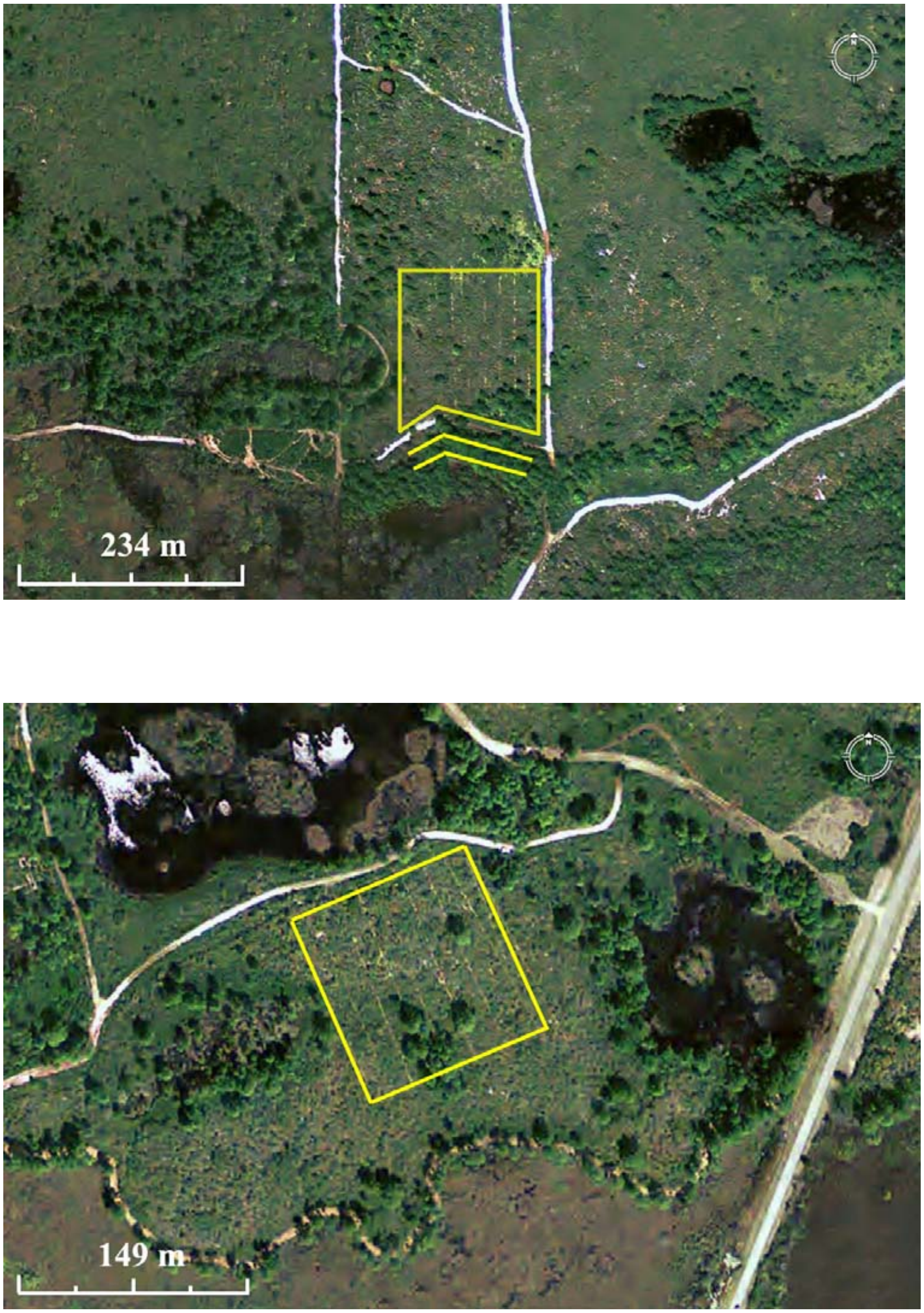
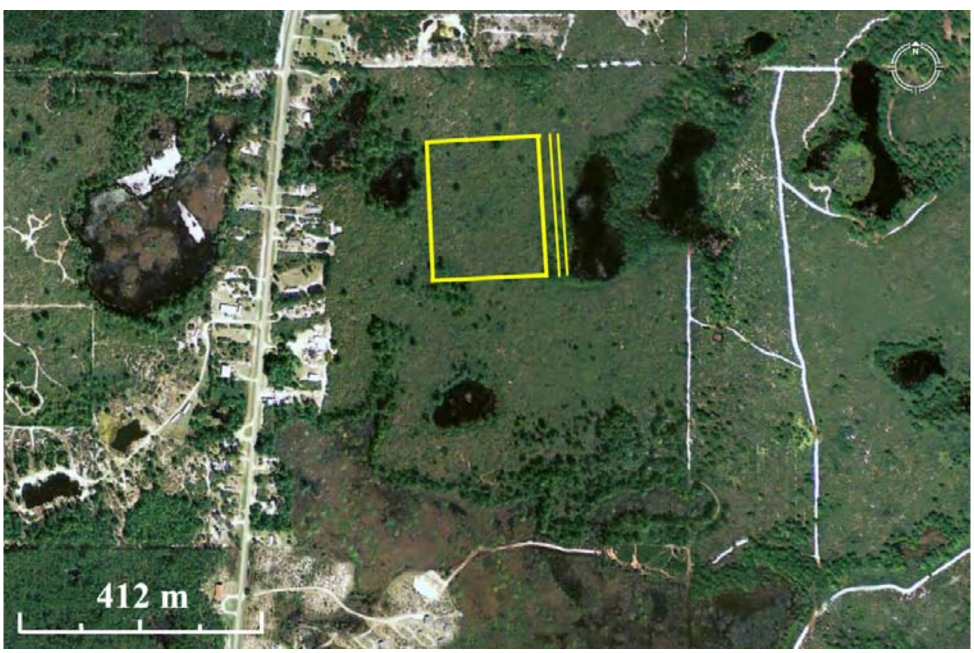


Figure 1-11. Treatment sites in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M. There were two transects of traps between grid and the wetland.

A



B

Figure 1-12. Control sites in Cedar Key Scrub State Reserve. A) Site 5A. There were two transects of traps between grid and the wetland. B) Site 5D.

CHAPTER 2
PRESCRIBED BURNING PLAN

Introduction

The prescribed burn plan for sites 5C and 2M was elaborated by the Park Manager Jeff DiMaggio. The prescribed burn plan considered the most relevant environmental variables that affect fire behavior and other factors of interest such as area to be burned, fire history, plant communities, smoke screening test, smoke sensitive areas, fire break/site preparation, special precaution for specific areas, firing procedure, contacted agencies, burn zone map, weather data, preburn checklist, safety procedures, and required personel and equipment. This plan required inspection of the sites to be burned in order to observe the condition of the vegetation (not previously burned or roller chopped) and to monitor environmental variables and the Keetch Byram Drought Index (KBDI) index (Keeth and Byram 1968) to assure that prescribed burning would be conducted under preferred environmental conditions. In addition, prescribed burning should be the same across treatment sites.

Fire behavior characteristics should not vary between treatments. This is a very strong constraint from the experimental point of view. Even though this was a difficult challenge to achieve, the park manager developed two plans with the same objectives and outcomes expected from them.

Objectives

The prescribed burning plan for sites 5C and 2M had the following objectives: (a) reduce vegetative biomass on the ground by a minimum of 60%, (b) top kill woody vegetation by a minimum of 75%, and (c) reduce the vegetative mass for habitat improvement for listed species.

This dissertation also had the following objectives: (a) to quantify rate of spread and flame length, (b) to indirectly estimate fire intensity through recording temperature during prescribed burning, (c) to determine if fire behavior characteristics were the same in treatment sites, and (d) to compare the results from Cedar Key fires with the predicted fire behavior from three model fuels in BehavePlus 3.0.2.

Methods

Before Burning

Mowing vegetation at the border of stands was the first site preparation (Figure 2-1). This job was carried out by using a Gyrotrac machine with a drum mounted on the front with many cutting blades. The border was mowed up to 3 m width around the stand and at least 2 months in advance of burning. In addition, a Brown tree cutter machine cut the remaining scrub to mineral soil to improve the firebreak.

Placing markers, taking vegetation samples, and installing sensors were needed to quantify rate of spread, moisture content, and temperature during prescribed burning, respectively. Existing pine trees, posts, and flags were used to mark specific places for quantifying distance and time from the starting ignition point. These distances and times were used to calculate rate of spread. A stratified random sampling was used to take vegetation samples in 20 points the day before burning between 11:00 am and 4:00 pm. Even though sampling points were assigned randomly, samples were selected in relatively undisturbed places and representative of the fuel complex and species composition. In each sampling point, a total of 20 samples of dry fuel were collected for each of the following classes: 1-h timelag (< ¼ inch diameter), 10-h timelag (1/4 to 1 inch), 100-h timelag (1 to 3 inches). In addition, 20 samples of live herb and woody vegetation were collected including only stem, branches, and leaves. Samples were put in brown paper bags, labeled, and weighed *in situ*. Later, samples were oven-dried for 10 days at 60 °C and reweighed. Fuel moisture content was calculated by using Equation 2-1:

$$\text{Percent moisture content} = \frac{(\text{Wet weight} - \text{Dry w}}{\text{Dry weight}} \times 100 \qquad (2\text{-}1)$$

At the same sampling point, two temperature sensors (Figure 2-2) were installed to measure temperature during burning. Mica tags (12.7 x 7.6 cm; material resistant to high temperature) were painted with five Tempilaq temperature-sensitive paints that melted at the following temperatures: 204 °C, 427 °C, 621 °C, 816 °C, and 1093 °C (see Figure 2-3). One tag was inserted at the ground level and the other was attached with wire to a tree at 1.5 m above the ground.

Details about the prescribed burning plan in 5C and 2M are displayed in Table 2-1. According to this table, the plan was the same for both sites. The differences between 5C and 2M prescribed burning plans were the area to be burned, the estimated flame length, and personnel needed.

Measurements During Prescribed Burning

The prescribed burning map for site 5C is presented in Figure 2-4. This map illustrates the ignition point (NE corner of the site) and where the fire lines ended (SW corner of the site). Fire line is the fire spread by the firefighter at intervals of approximately 15-20 m at the border of the site in order to control the spread of the fire (Figure 2-5). The test fire was completed at 10:38 am, and two fire lines were immediately started at the border of the site. The first line started on the north side of the site, moved west and later moved south on the west side of the stand. The second line began at the same point where the first line was started and moved south on the east side of the site. The two lines met at the SW corner of the stand. This burning design was planned to burn the site in an effective way.

Burning with a combination of back and head fire made the plan successful. The park manager knew in advance from previous days that wind was blowing mainly in the SE-SW direction and planned to start burning at the SW corner of the site. However, wind direction shifted to NW-W the evening before burning and in the morning of the burning day. Therefore, the park manager decided to start burning at the NE corner due to the shift in wind direction and because 5C should start burning with back fire. The two fire lines moved simultaneously, but the first line burned faster than the second one. Then, the park manager coordinated the movement of fire lines on both the west and east sides of the stand in such a way that they did not move farther than half site 5C going south. The idea was to burn half 5C at the north side first by using head fire from the west side fire line (moving south on the west side of 5C) and back fire from the east side fire line (Figure 2-6). In this way, we created a “firebreak” for the head fire originated by the time the fire lines were near the SW corner of the stand. This plan worked even though wind direction changed 14 times between 10:38 am and 2:09 pm. By the last time, 5C was completely burned.

Prescribed burning in 2M used the same technique as in 5C. Wind direction was SE-SW on previous days, so the park manager decided to start at point A of 2M (Figure 2-4) in order to burn the treatment site first (where the trapping grid was located). Fire test was carried out at 10:54 am (Figure 2-7) and immediately after it, a first fire line started going NE at the border of 2M (Figure 2-4 and 2-7). From 10:54 am to 1:25 pm, the front fire (back fire) moved slowly and reached transect C of the trapping grid (approximately half the distance between the first and third yellow arrow at the border of 2M on Figure 2-4), and from 1:25 pm to 2:20 pm, it moved fast (head fire) because the wind changed direction to NE. This change in wind direction not only increased rate of spread, but also increased fire intensity. Burning the side of 2M where the trapping grid was located finished at 2:20 pm. Starting points B and C began at 11:20 am and 4:00 pm, respectively, and the corresponding fire lines burned the rest of 2M.

Environmental variables were quantified during prescribed burning every 30 minutes. A Dwyer hand-held wind meter was used to record wind speed, and a Sling-Psychrometer was used to measure air temperature and relative humidity. In addition, cloud type, state of weather, and fire conditions were monitored.

Photograph documentation and monitoring time of back and head fires were carried out during prescribed burning. A digital camera was used to take pictures and mini videos (up to 3 minutes) during the entire process. Since these pictures would be used to estimate flame length, an object or a firefighter was used as a reference (Figure 2-8). Out of 92 and 79 pictures taken in 5C and 2M, respectively, 30 pictures were selected in each site to measure flame length. Rate of spread of the fire front was measured from the time the firefighter started to make the fire line. Special attention was focused on changes in wind direction in order to quantify back or head fire rate of spread. Back and head fires were only quantified up to 20-30 m from the border of the site because the visibility was limited due to smoke. Standing at the top of a truck helped me to watch and to record the advance of the fire front until it reached marked pine trees, posts, or flags.

Measurements After Prescribed Burning

Sites 5C and 2M were monitored right after prescribed burning. Firefighters reviewed each site after burning, particularly 8-10 m from the border of each site in order to look for spots still burning and to stop these fires by adding water. In addition, firefighters also sprayed water on pine trees still burning. The burned areas were re-checked during the night and the next day for smoke and flare-ups. I visited sites 5C and 2M the next day after prescribed burning with the purpose of collecting temperature sensors, estimating flame length with the pictures *in situ*, taking pictures, and looking for wildlife killed by fire.

Predicting Fire Behavior

The program BehavePlus 3.0.2 (Andrews and Bevins 2005) was used to predict fire behavior by using the default worksheet. This worksheet contains fuel models, fuel moisture, surface wind speed, and slope steepness. Fuel model 4 (Chaparral), fuel model sh5 (high load, dry climate shrub), and fuel model sh8 (high load, humid climate shrub) were run with the moisture contents measured for the three types of dry fuel, live herbs, and live woody vegetation collected before burning and the wind speed recorded during burning. Slope steepness was input to zero. Fuel model 4 was selected because this is the model for the shrub group characterized by the California mixed chaparral. Fuel model sh5 was chosen because both sites had low precipitation during the last 2 weeks before burning (91.1 mm in April and 93.6 mm in May). However, model sh8 was also selected because the KBDI index during prescribed burning (5C = 220 and 2M = 234) suggested soils had wet conditions. The minimum, maximum, and average wind speed registered during prescribed burning and the average fuel moisture content for each dry and live fuel type were input into each model. As a consequence, each model produced three rates of spread and flame lengths. These results were compared with the minimum, maximum, and average rates of spread and flame length observed during prescribed burning and calculated from the distance/time quantified in situ for head fire and from the pictures taken during prescribed burning.

Results and Discussion

Rate of Spread, Flame Length, and Fire Intensity

Table 2-2 shows duration of prescribed burning, wind direction change, surface wind speed, air temperature, air relative humidity, fuel moisture content, and KBDI index during prescribed burning in 5C and 2M. The duration of the burning was approximately the same in both sites. However, the time recorded for 2M corresponded only to the burning time for the portion of 2M where the trapping grid was installed. Wind changed direction 14 times in 5C and one time in 2M. This un-controlled variable affected fire behavior in 5C and the difference between the two sites regarding how burning occurred in both sites. Wind speeds were not significantly different between the two sites, though, but air temperature and air relative humidity were significantly higher in 2M than in 5C (Appendix A). Average fuel moisture content for each fuel type was slightly higher in 2M than in 5C with the exception of live herbaceous, but they were not significantly different (Appendix A). Probably, this was due to the higher air temperature and relative humidity found in 2M. The KBDI index suggested wet soils in both sites by the time of the burning.

Observed fire behavior characteristics such as flame length, rate of spread, and fire intensity are presented on Table 2-3. Flame length and rate of spread were slightly higher in 5C than in 2M, but they were not significantly different (Appendix A). Average fire intensity was significantly lower in 5C than in 2M. The change in wind direction probably was one of the factors that varied and lowered fire intensity in 5C (Appendix A). In this site, out of 60 censors, the temperature registered by 28 censors varied between 204 °C and 816 °C, and the rest of the censors experienced temperatures equal or higher than 1093 °C. I state that the temperature was higher than 1093 °C because there was no paint remaining on the mica sheets. In 2M, out of 60 censors, five censors recorded 204 °C, five censors registered 427 °C, one censor recorded 816

°C, and the rest experienced temperatures equal to or higher than 1093 °C. So, fire intensity was higher an almost homogeneous in 2M in comparison with 5C. Therefore, fire behavior characteristics were not exactly the same in treatment sites, but at least flame length and rate of spread were similar. Even though fire intensity was not the same in treatment sites at the heights where tempetature was recorded, it was high enough to reduce almost 100% of the vegetation and top kill almost 100% of the above-ground woody vegetation in both sites with very little damage to wildlife.

The objectives of the prescribed burning plans were achieved. Figures 2-9 through 2-11 illustrate how sites 5C and 2M appeared the day after burning. Almost all trees were burned in 5C and all trees were burned in 2M. Since fire intensity varied in 5C, flames did not completely consume all tree foliage. Approximately, 7% of the trees did not burn completely, but the stems fell down after several weeks. Also, I only found two *Terrapene carolina bauri* (eastern box turtle) in 5C, one in 2M, and several insects burned after prescribed burning (Figures 2-12).

Fuel Model Predictions

The comparison among models and the results found in Cedar Key are shown in Figures 2-13 and 2-14. A clarification about this comparison is needed because models predicted values according to mathematical models, but this is not the case for the relationship between observed rate of spread / flame length and surface wind speed in Cedar Key. The minimum, average, and maximum observed values of wind speed were matched with the minimum, average, and maximum observed values for rate of spread and flame length. This match was a valid assumption because it was expected that minimum, average, and maximum rates of spread or flame length would occur when wind speed was also at the minimum, average, and maximum value. Based on this assumption, the comparison was made. Another assumption was that surface wind speed measured during prescribed burning was equal to midflame wind speed used in BehavePlus.

BehavePlus models a nearly linear relationship between rate of spread or flame length with midflame wind speed. In both sites, rate of spread or flame length increased when wind speed increased. In addition, rate of spread or flame length decreased when relative humidity increased. However, more data are needed to confirm the linear relationship found in this study. Figure 2-13 illustrates that rates of spreads found in Cedar Key fell between models sh8 and sh5 or between humid and dry climate shrubs in both sites. It is important to highlight that the values from Cedar Key were close to the predicted values from model sh8 in 2M. Model 4 and sh5 over-estimated rate of spread and flame length for Cedar Key. Figure 2-14 presents a different scenario. The average flame length recorded in Cedar Key was between model sh5 and model 4, but the minimum and maximum values of flame length fell beyond the predicted values from the models in both sites. Now, the reality was that minimum and maximum flame lengths were obtained from pictures taken during prescribed burning in both sites. This is a fact illustrated on Figure 2-15. Therefore, models 4, sh5, and sh8 over-estimated flame length at the minimum wind speed and under-estimated flame length at the maximum wind speed recorded during prescribed burning in both sites.

Prescribed burning studies have not reported fire behavior characteristics in the reviewed literature. Few studies have described the environmental variables during prescribed burning (Abrahamson and Abrahamson 1996a, Weekly and Menges 2003, and Greenberg 2003); however, fire behavior characteristics were not reported. This is a critical aspect in experimental designs that focus in fire effects. Although it is difficult to achieve the same fire behavior characteristics among sites considered under treatment, an attempt must be made in order to know how similar/dissimilar stands are regarding the application of the treatment.

Table 2-1. Comparison of the prescribed burning plan between sites 5C and 2M in Cedar Key Scrub State Reserve.

Category	5C	2M
Area to burn	12 ha	33 ha
Starting time	10:00 AM	10:00 AM
Last burn/years fire suppression	1955 / 50 years	1955 / 50 years
Fire procedure	Baking, flanking, strip head	Baking, flanking, strip head
Wind direction	SE-SW	SE-SW
Surface wind speed (Min/Max)	11 / 23 kph	11 / 23 kph
Transport wind speed (Min / Max)	15 / 32 kph	15 / 32 kph
Minimum mixing height	610 m	610 m
Dispersion Index	Day 65 max	Day 65 max
Air temperature (Min / Max)	4 / 32 °C	4 / 32 °C
Air relative humidity maximum (Min / Max)	35-50 %	35-50 %
Fine fuel moisture content	8-12%	8-12%
Drought index	< 550	< 550
Rate of spread	2-5 m/min	2-5 m/min
Flame length	2-7 m	2-5 m
Personnel required	6-8	6-10
Passed smoke screening system	Yes	Yes

Table 2-2. Environmental variables during prescribed burning in sites 5C and 2M in Cedar Key Scrub State Reserve.

Category	Category	5C	2M
Date		04/21/05	05/18/05
Starting time		10:38	10:54
Ending time		14:09	14:20
Duration		3:31 h	3:26 h
Wind direction change		14 times	1 time
Surface wind speed (kph)	Minimum	2.41	1.61
	Maximum	4.83	4.83
	Average	3.22	3.62
Air temperature (°C)	Minimum	23.89	26.67
	Maximum	27.22	32.22
	Average	26.60	29.31
Air relative humidity (%)	Minimum	40.00	55.00
	Maximum	54.00	75.00
	Average	49.00	62.00
FMC 1h timelag (%)	Minimum	3.81	2.43
	Maximum	10.10	11.96
	Average	6.51	7.42
FMC 10h timelag (%)	Minimum	1.69	3.45
	Maximum	19.27	22.20
	Average	7.52	8.31
FMC 100h timelag (%)	Minimum	6.08	4.41
	Maximum	36.14	38.85
	Average	13.95	14.85
FMC live herb (%)	Minimum	27.36	19.66
	Maximum	86.92	82.93
	Average	59.20	55.93
FMC live woody (%)	Minimum	42.86	39.48
	Maximum	86.05	74.94
	Average	60.56	62.13
KBDI index		220	234

Codes: FMC = fuel moisture content. KBDI = Keetch Byram Drought Index.

Table 2-3. Observed fire behavior characteristics in sites 5C and 2M in Cedar Key Scrub State Reserve.

Category		5C	2M
Flame length (m)	Minimum	1.00	1.00
	Maximum	6.50	5.50
	Average	3.99	3.44
Rate of spread-back fire (m/min)	Minimum	0.78	0.73
	Maximum	2.29	2.22
	Average	1.66	1.47
Rate of spread-head fire (m/min)	Minimum	3.71	2.22
	Maximum	8.67	6.67
	Average	6.21	4.87
Fire intensity-Temperature at the ground level (°C)	Minimum	204.0	204.0
	Maximum	1093.0	1093.0
	Average	858.0	1019.0
Fire intensity-Temperature at 1.5 m above the ground level (°C)	Minimum	204.0	204.0
	Maximum	1093.0	1093.0
	Average	600.0	899.0

Figure 2-1. Improvement of the firebreak located at the north side of site 2M in Cedar Key Scrub State Reserve. The road is the fire break and it can be seen at the left side (6 cm from left to right) of the picture. The firebreak was improved by increasing its width (the remaining 6 cm from middle to the right of the picture).

A

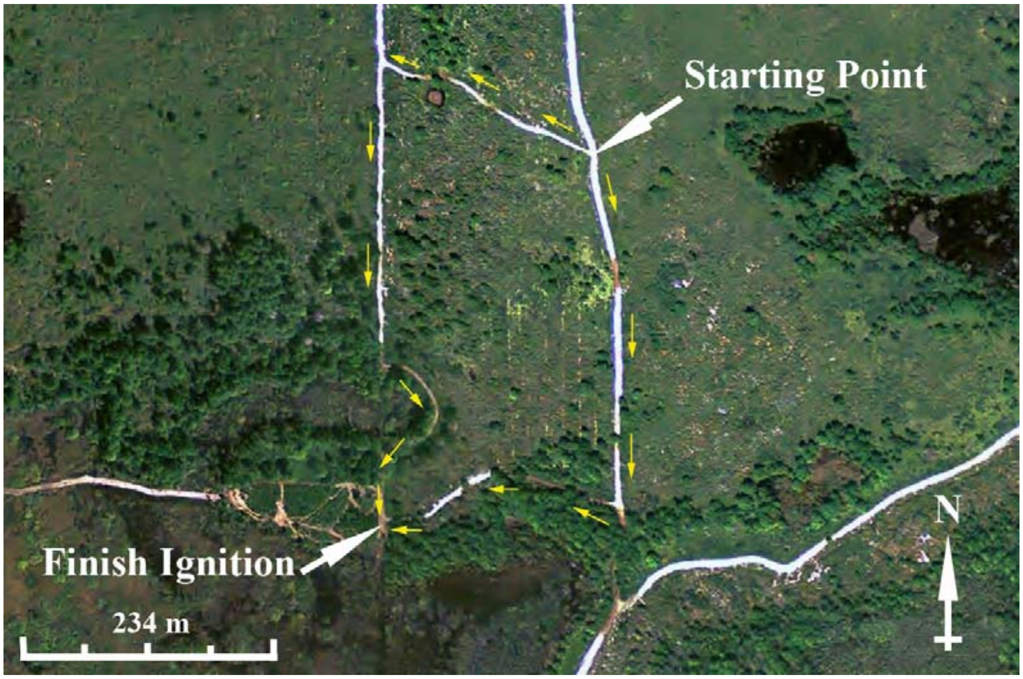
B

Figure 2-2. Sensor with bands of temperature-sensitive painting. A) Sensor not exposed to fire. B) First band was melted indicating that temperature reached 204 °C.

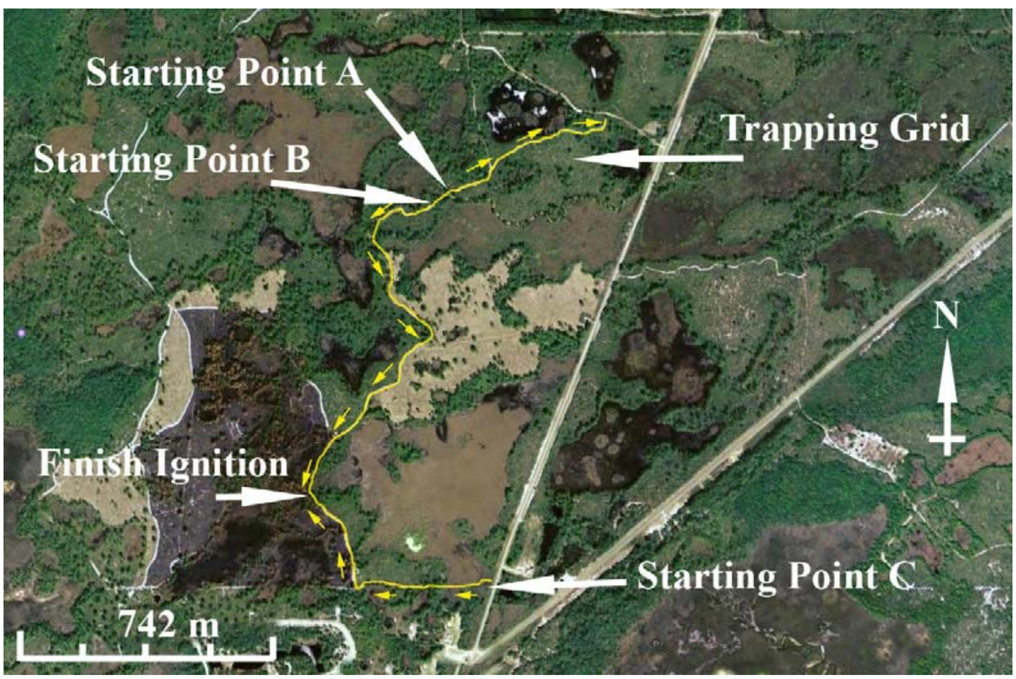
A

B

Figure 2-3. Sensors experimented temperature ≥ 1094 °C. A) Sensor at 1.5 m above the ground. B) Sensor at the ground level.



A



B

Figure 2-4. Prescribe burning maps in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M

Figure 2-5. Fire line created by the firefighter at the border of the stand in Cedar Key Scrub State Reserve.


A


B

Figure 2-6. Examples of fire front in Cedar Key Scrub State Reserve. A) Back fire. B) Head fire.

A

B

Figure 2-7. Applying prescribed burning in Cedar Key Scrub State Reserve. A) Fire test in site 2M. B) Fire line at the border of the site 2M.

A

B

Figure 2-8. Taking pictures with a reference person of know height in Cedar Key Scrub State Reserve. A) Jeff DiMaggio in 5C. B) David Romano in 2M.

A

B

Figure 2-9. Effect of prescribed burning in the scrubby flatwoods in site 5C in Cedar Key Scrub State Reserve. A) Before burning. B) After burning.

A

B

Figure 2-10. Effect of prescribed burning in the scrubby flatwoods in site 2M in Cedar Key Scrub State Reserve. A) Before burning. B) After burning.

A

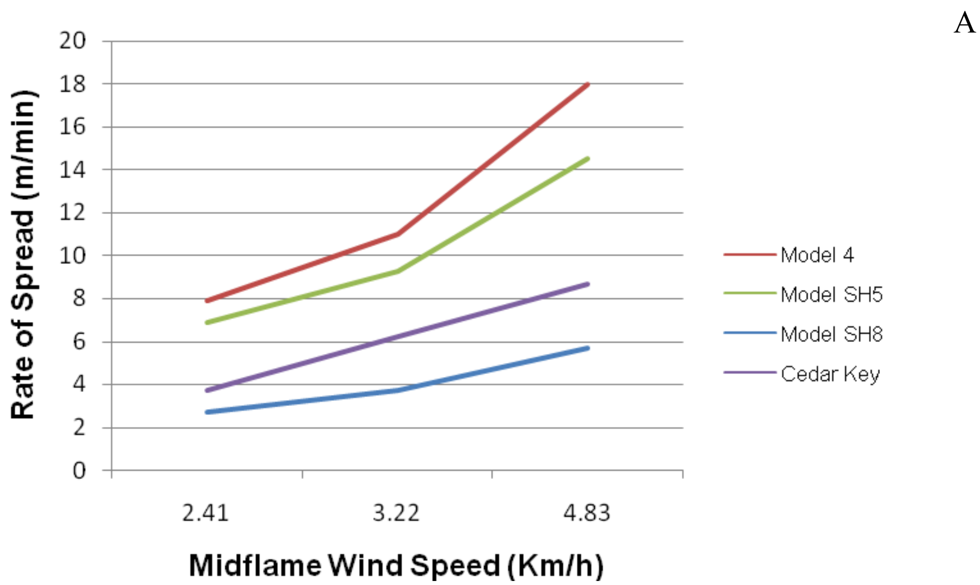
B

Figure 2-11. After prescribed burning in Cedar Key Scrub State Reserve. A) Site 5C showing a burned palm, palmettos, and pine trees. B) Site 2M with burned oaks and palmettos.

A

B

Figure 2-12. Burned animals in Cedar Key Scrub State Reserve. A) *Terrapene carolina bauri* (eastern box turtle). B) Cockroaches.



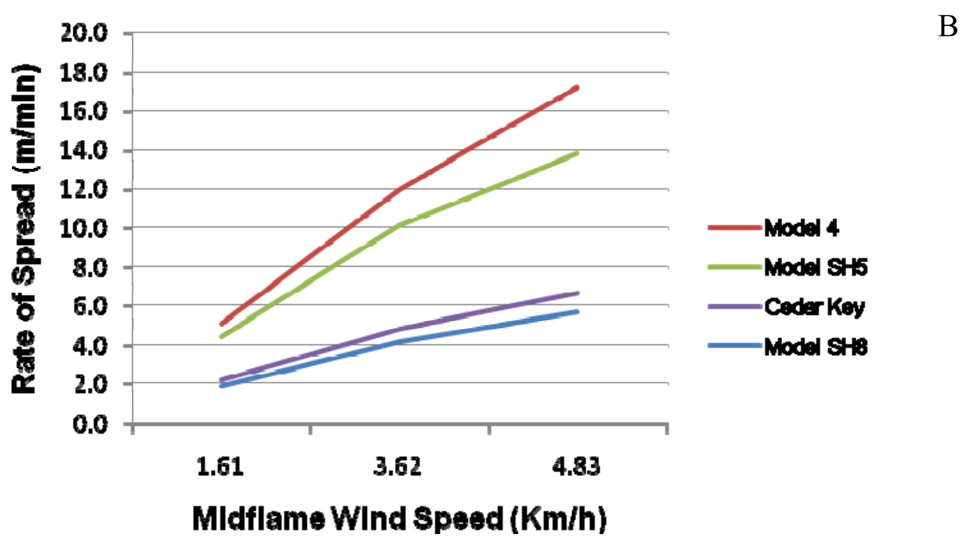
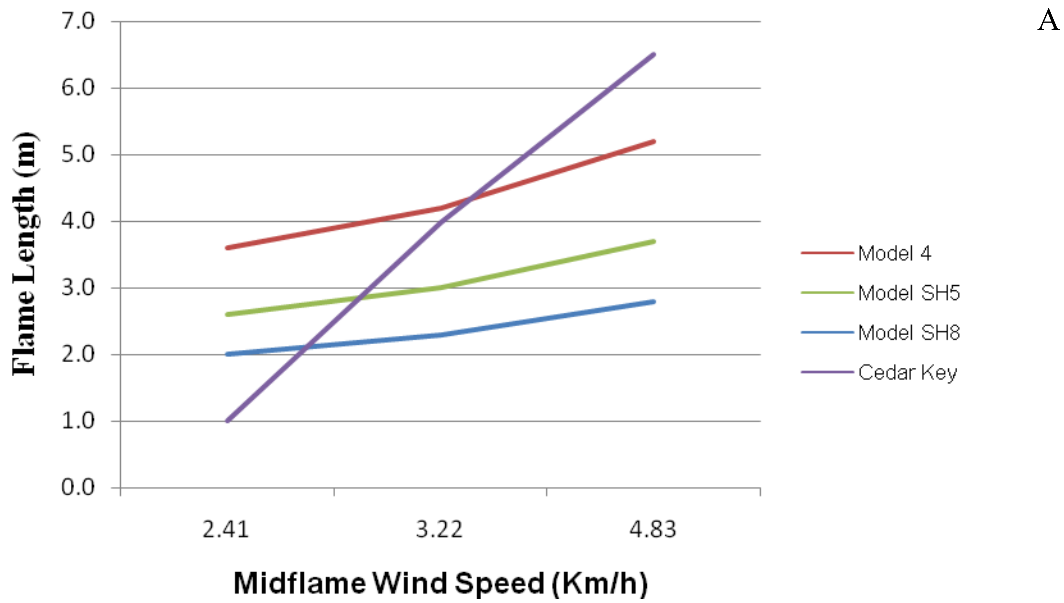


Figure 2-13. Rate of spread comparison between Cedar Key and fuel models 4, sh5, and sh8 according to BehavePlus 3.0.2. A) Site 5C. B) Site 2M.



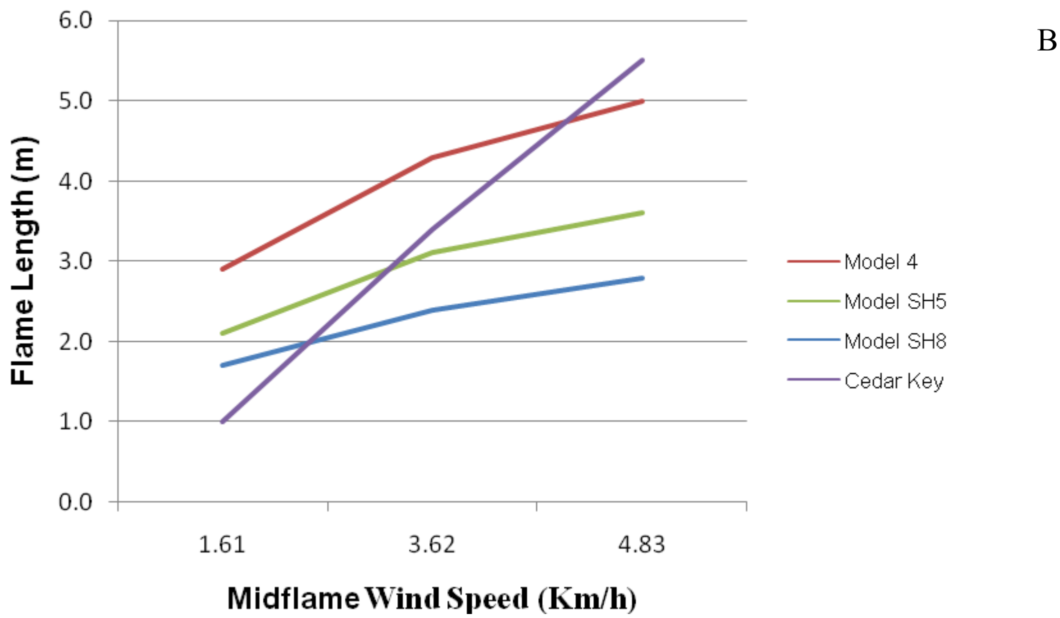


Figure 2-14. Flame length comparison between Cedar Key and fuel models 4, sh5, and sh8 according to BehavePlus 3.0.2. A) Site 5C. B) Site 2M.

A

B

Figure 2-15. Examples of flame length in Cedar Key Scrub State Reserve. A) Flame length up to one m height. B) Flame length of at least 6 m height.

CHAPTER 3
RESPONSES OF LONG-UNBURNED SCRUBBY FLATWOODS TO BURNING

Introduction

Florida has had colossal changes that affected the size and the distribution of the scrub ecosystem. The scrub habitat probably appeared during the early Tertiary Period originating in the southern Rocky Mountains and northern Mexico with eventual spread along the Gulf coast to Florida (Axelrod 1958). In Florida, the scrub habitat formed during the early Miocene (20 million years before the present time (mybp)), and it is one of the most ancestral habitats of Florida in conjunction with the mesic forest (Webb 1990). During the Pliocene epoch, the Florida peninsula started a process of contraction and expansion of land area because of sea level rise during glacial and interglacial periods. At one time, Florida was about two times the current size because of lower sea level and xeric conditions. During the late-Pleistocene, the scrub vegetation was probably wide-spread across the peninsula. However, during the later part of this epoch, sea level rose and reduced the size of the Florida Peninsula, and mesic conditions extended into the landscape (Myers 1990). During the last million years (still the Pleistocene), broad areas of xeric habitats persisted in Florida, but they were replaced by wet subtropical (mesic) habitat because of increased precipitation and increased water tables. In this process, the original widespread and almost continuous scrub habitat was reduced and became fragmented (Webb 1974, Clark et al. 1999). The fragmentation process increased in the last 5000 – 7000 years because the climate became more humid, water levels rose, and electrical storms and lightning fires developed. In addition, natural fragmentation has been coupled with human perturbations in the last 200 years.

The current distribution of the Florida scrub is a consequence of the historical biogeography of the Florida peninsula plus the effect of human perturbations. During the post-

European period, the natural fragmented scrub ecosystem was gradually reduced by conversion to housing developments, citrus groves, and golf courses (Myers 1990). From 1940 to 1981, approximately 64% of the xeric upland habitat was destroyed and an additional 10% was disturbed (Peroni and Abrahamson 1985). The scrubs have lost more than 60% of the original extent, and 85% of the scrubs in Lake Wales Ridge were converted to agriculture, commercial, or residential development (Peroni and Abrahamson 1985, Christman 1988). The last five decades of development have reduced considerably the extent of the scrubs (McCoy and Mushinsky 1994). These perturbations have decreased the number and size of scrub patches and have increased their isolation. As a result, the original widespread scrub ecosystem has been fragmented and reduced to patches in the interior and along the coast of the peninsula.

Scrubs in Florida have different ages. Because of the glacial and interglacial periods, at least six ancient shorelines were created during the 25 mybp, and the scrub habitats along the central portion of Florida are the oldest (approximately 9 mybp) among all scrub habitats (Myers 1990). So, the current Florida landscape can be described as old scrub habitat on the north – south axis in the center of the state, and young scrub habitats towards the coastlines (approximately 0.5-2 mybp). Also, the scrub habitat can be classified as inland or coastal scrub. Inland scrubs are the largest block of scrubs and occur along a complex of sand ridges running north-south from Clay and Putnam Counties to Highland County. These sand ridges, dated from the Miocene to early Pleistocene, form the Florida Central Ridge, and they include: Ocala National Forest, Lake Wales Ridge (conformed by Arbuckle, Carter Creek, and Archbold) and Avon Park Air Force Range (Myers 1990, Clark et al. 1999). These ridges have been separated by several kilometers, and each ridge has an assemblage of scrub patches separated by mesic habitats and by human development that together act as a matrix for scrub species. Coastal scrubs are the smallest scrubs found on both the Atlantic and Gulf coasts. The northernmost examples of these scrubs are located in the Panhandle and are restricted to a narrow strip along the Gulf coast. They extend from west Ochlockonee River in Franklin County, Florida, to Gulf Bay State Park in Baldwin County, Alabama. In north-central Florida, coastal scrubs occur on the east coast in St. John's County near Durbin and on the west coast in Levy County near Cedar Key. The southernmost scrubs are found on the west coast at Marco Island in Collier County (probably already extirpated) and on the east coast in Merritt Island National Wildlife Refuge, Cape Canaveral barrier island complex (Kennedy Space Center), and Jonathan Dickinson State Park. A high number of endemic species, which are habitat specialists, characterize inland and coastal scrubs.

Scrub is the most unique and restricted natural ecosystem in Florida. The scrub in Florida is a shrubland ecosystem located on contemporary or relict beach dune substrates maintained by recurrent disturbances (Myers 1990, Gibson and Menges 1995, Menges 1999). Scrub communities are dominated by a well-developed layer of evergreen oaks (shrubs), with or without a sand pine overstory, sparse ground cover with few herbaceous plants, and many patches of bare ground occupying well-drained, infertile, sandy soils (Layne 1963, 1990; Myers 1990). Oaks species typically include *Quercus geminata*, *Q. myrtifolia*, *Q. inopina* (inopine oak) and *Q. chapmanii*. In addition, other shrubs such as *Lyonia ferruginea* and *Ceratiola ericoides* (false rosemary) are common. Ninety percent of the shrub layer consists of the same six species in approximately the same order of abundance: *Q. myrtifolia*, *Q. inopina*, *Serenoa repens*, *L. ferruginea*, and *C. ericoides* (Myers 1990). The ground cover is always sparse and includes species such as *Cladonia evansii* (deer moss), *Licania michauxii* (gopher apple), *Galactia* spp. (milk peas), and other herbs.

Several types of scrub have been named depending on dominant species, location, elevation, soil type, fire history, and other factors (Myers 1990, Menge 1999). These scrubs are rosemary scrub, oak scrub, oak-saw palmetto scrub, sand pine scrub, slash pine scrub, and scrubby flatwoods. Rosemary scrub is the most common, and it is characterized by the common species *C. ericoides* and by gaps supporting an herbaceous flora that includes terrestrial lichens and many rare species. Oak scrub and oak-saw palmetto scrub are dominated by oaks and the association oak-saw palmetto, respectively. Sand pine and slash pine scrubs have sand pine and slash pine, respectively, as representative species in conjunction with other shrub species. The name scrubby flatwoods is applied to scrubs that either lack a pine overstory or have slash pine in place of sand pine. Scrub is one of the most endangered communities in Florida not only for the number, size, and distribution of the patches, but also because of the number of endemic plant and animal species.

Scrubs have several endemic plant and animal species. Currently, 22 plant species are federally-listed as endangered or threatened (U.S. Fish and Wildlife Service 1999). Examples of these are *Ilex opaca* (scrub holly), *Persea humilis* (silk bay), *Garberia heterophylla* (garberia), *Palafoxia feayi* (palafoxia), and *Osmanthus megacarpa* (wild olive). However, the truly rare endemic species are restricted to the Lake Wales Ridges such as *Hypericum cumulicola* (scrub hypericum), *Dicerandra frutescens* (scrub balm), *Eryngium cuneifolium* (wedge-leave snakeroot), *Lupinus aridorum* (Beckner’s lupine), and *Warea carteri* (Carter’s warea). This concentration of endemism is probably due to the age of the scrub and its isolation. A very rare shrub is *Ziziphus celata* (Garret’s ziziphus), collected only twice, not seen since 1955, but rediscovered in 1987 (Myers 1990). Vertebrate endemic species that occur in the scrub are the following: *Podomys floridanus* (Florida mouse), *Aphelocoma caerulecens* (Florida scrub jay), *Sceloropus woodi* (Florida scrub lizard), *Neoseps reynoldsi* (sand skink), and a mole skink species with three subspecies, *Eumeces egrerius egrerius* (brown red-tailed skink), *E. e. lividus* (blue-tailed skink), and *E. e. insularis*. The structure and stage of the vegetation is very important for these species and others. For example, if the height of the scrub reaches a critical level, and a pine canopy develops, then *P. floridanus*, *A. caerulecens*, and many bird species leave the patch of scrub (Myers 1990). In addition, the development of a mature scrub creates habitat for other species such as *Glaucomys volans* (flying squirrel), *Sciurus carolenensis* (gray squirrel), *Ochrotomys nuttalli* (golden mouse), *Peromyscus gossypinus* (cotton mouse), and many species of birds. Therefore, scrub species need a very specific structure and stage of the vegetation for habitat, and the only way that scrub communities maintain this status is through fire periodicity.

Scrub is a pyrogenic ecosystem that requires catastrophic fire for self-maintenance. Scrub fires are devastating, resulting in extensive consumption of the above ground vegetation. The natural frequency of fires is one every 10-100 years according to Myers (1990) or one every 20-50 years according to Layne (1990). In the past, the scrub ecosystem had a natural fire frequency that allowed its maintenance and persistence. In the absence of a natural frequency of fire, tree and shrub layer density increase and scrub transforms into xeric hardwood forest. However, when frequency of fire becomes more frequent, sand pine disappears, and the association becomes oak scrub or changes to high pine (Myers 1990). This was what actually happened during the pre- and post-European periods. Humans altered the natural frequency and intensity of fires. As a result, the scrub ecosystem become more fragmented or changed to another type of vegetation. Currently, prescribed burning is extensively used in Florida, and it has helped to maintain the scrub ecosystem. Under prescribed fire and natural conditions, fire maintains sand pine scrub and scrubby flatwoods (Layne 1990, Myers 1990) as stable and non-successional associations. How plant species respond to prescribed burning has been analyzed by several studies.

There are two general approaches to describe the effects of prescribed fire on flora. The first one takes into consideration the concept of fire regimen in order to understand fire effect at the community level. The second one focuses on the response at the species level. Let us starts with the definition of fire regimen.

The concept of fire regimen encompasses several components. Kilgore (1987) defines fire regimen as a set of several factors such as fire frequency (time between fires), season of burn, fire periodicity, fire intensity, size of fire, pattern on the landscape, and depth of burn. Brown and Smith’s (2000) definition includes pattern of fire occurrence, size, uniformity, and severity. Whelan (1995) considers fire regimen as a global concept to summarize fire frequency, season of burning, type of fire (only organic layer soil, only above ground, or crown fire) and extent of the fire (continuous vs patchy). This dissertation synthesizes the three previous concepts and defines fire regimen as a global concept to summarize fire frequency, intensity, severity, type of fire, size, season, and extent. Fire regimen has been used in order to categorize plant community responses to fire.

The most recent classification of plant community responses to fire uses fire severity as the main criterion. Brown and Smith (2000) used a fire regimen classification relying only on fire severity. According to those authors, the use of fire severity as the key component to describe fire regimen is interesting because it connects directly to the effects of disturbance, especially on survival and structure of the dominant vegetation. Brown and Smith’s (2000) classification is as follows:

- Understory fire regime (applies to forest and woodland vegetation). Approximately 80% or more of the aboveground dominant vegetation survive fire. Fire is not lethal for dominant vegetation and does not change its structure.

- Stand-replacement regime (applies to forests, woodlands, shrublands, and grasslands). Approximately 80% or more of the aboveground dominant vegetation is either consumed or killed by fires. Since fire consumed or killed the aboveground parts of the dominant vegetation, they dramatically change the structure of it as well, particularly in shrublands and forests.

- Mixed severity regime (applies to forests and woodlands). Fire selectively kills species of the dominant vegetation depending on the species' susceptibility to fire. This type of fire varies between understory and stand-replacement.

- Nonfire regime. Little or no occurrence of natural fire. All type of forest can be classified according to the categories above that correspond to low, medium, and high fire severity types.

The second general approach about how plant species respond to prescribed burning takes into consideration how species survive fire. Whelan (1995) has named four categories as follows: fire ephemerals, obligate seeders, sprouters, and facultative sprouters. The first category describes plants that do not survive the fire. The second category refers to plants that germinate after fire through a seed bank stored in the soil or in the canopy. The third category presents plant species that survive fire through protected buds in the stems or roots. The fourth category introduces species in which the ability to sprout would depend on the characteristics of the prescribed fire. However, recovery mode of individual species in Florida scrubs may be correlated with habitat characteristics and fire regime.

The scrub ecosystem in Florida falls under the category of a stand-replacement regime and the majority of the species are resprouters. After a long fire-free period of fuel accumulation, a high intensity fire takes place. If sand pine or slash pines are present, they might be killed. The above-ground shrub layer is totally consumed. According to Menges and Kohfeldt (1995), species recovery varies, and they classified 95 species of the scrubby flatwoods and rosemary scrub into seven guilds of recovery mechanism: 24 species were resprouters (many woody shrubs), 24 were resprouters and seeders (small-statured shrubs, palmettos, and herbaceous perennials), 14 were resprouters and clonal spreaders (the majority of the dominant shrub genera in scrubby flatwoods such as *Quercus*, *Lyonia* , and *Vaccinium*), five were resprouters, clonal spreaders, and seeders (herbs), 26 were obligate seeders (*C. ericoides* and many herbs), one aerial seeder (*Pinus clausa*), and one seeder and survivor (*Pinus elliottii*). Shrub species sprout from previously suppressed underground buds on buried roots. Few shrub species, for instance *C. ericoides*, regenerate from seeds stored in the soil. *P. clausa* regenerates from fire-induced seed release from individuals with serotinous closed cones. *P. elliottii* is the only pine tree that might survive moderate or high-intensity fire and also recovers by seeds. Both species might reseed from other pine trees in adjacent stands. Post-fire species composition is usually an assemblage of many of the species previously growing on the site. However, there are some variations in the way Florida scrubs recover after fire.

Florida inland scrubs, such as scrubby flatwoods and rosemary scrubs, have different recovery strategies (Menge and Kohfeldt 1995). Scrubby flatwoods specialists usually depend on vegetative recovery modes (61%; resprouting and clonal spread) and less often on mixed modes (23%) or obligate seeding (16%). In general, specialist species and dominant shrubs spread clonally and resprout in the scrubby flatwoods. In contrast, half (50%) of the of rosemary scrub specialists are obligate seeders, 17% were mixed, and 33% were vegetative. Also, the dominant species of *C. ericoides* are mainly obligate seeders. Species found in both habitats are intermediate in recovery modes (32% vegetative, 36% mixed, 32% obligate seeders). Therefore, in general, scrubby flatwoods appear to be more favorable for post-fire resprouting and clonal growth and rosemary scrub more favorable for post-fire seedling recovery. This difference occurs despite the overlap in species composition (Abrahamson et al. 1984) and close concurrence of these communities in the landscape. Coexistence of resprouters, seeders, and species with mixed recovery modes in Florida inland scrubs suggests that fire-return intervals may be quite variable.

Post-fire recovery of Florida inland scrub ecosystem varies with dominant shrub species and fire history. Most scrubby flatwoods species recover by resprouting and/or clonal spread oaks, recovery is rapid, and there is little change in species composition at a scale of four to 10 years (Abrahamson 1984a, 1984b, Johnson and Abrahamson 1990, Abrahamson and Abrahamson 1996b). In contrast, recovery of rosemary scrub takes more time because *C. ericoides* and *P. clausa* (the dominant species) recover through post-fire seedling establishment, and *C. ericoides* takes a decade to reach sexual maturity (Johnson 1982, Johnson et al. 1986). Fire return intervals for scrubby flatwoods vary from 5 to 20 years (Menges & Kohfeldt 1995) and to 20 to 80 years for rosemary scrubs (Myers 1990, Menges 1999).

Post-fire recovery of Florida coastal scrub also varies with dominant species and fire history. The majority of plant studies conducted in coastal scrub have been carried out on Merritt Island National Wildlife Refuge and Cape Canaveral (Simon 1986, Breininger and Schmalzer 1990, Schmalzer and Hinkle 1991, 1992a, 1992b, Schmalzer and Boyle 1998, Schmalzer and Adrian 2001, Schmalzer 2003, Schmalzer et al. 2003). Myers (1990) classified this scrub as a coastal scrub; Weekly and Menges (2003) refered to it as a coastal oak-palmetto scrub. However, according to Schmalzer et al. (2003), the most common types of plant communities in Merritt Island and Cape Canaveral are oak-saw palmetto scrub, scrubby flatwoods, and coastal scrub. Schmalzer (2003) classified the scrubby flatwoods without slash pine overstory as oak-saw palmetto scrub**.** Dominant species in these two communities were: myrtle oak, sand live oak, Chapman oak, saw palmetto, and ericaceous shrubs such as *L. ferruginea*. Schmalzer et al.

(2003) considered coastal scrub a different type of community because it was dominated by *Quercus virginiana* (Live Oak), *Serenoa repens*, and ericaceous shrub species were absent. The type of soils was also different. Oak-saw palmetto scrub was on soils that varied from neutral to acid, but the majority was acid soils. Coastal scrub soils were alkaline. In oak-saw palmetto scrub, recovery of dominant oaks and ericaceous species after fire is primarily through resprouting and clonal spread (Schmalzer and Hinkle 1992a, 1992b). Resprouting allowed a rapid reestablishment of the dominant shrubs. Saw-palmetto reestablished cover faster than woody shrubs. Saw palmetto cover equaled preburn values between one and 1.5 year postburn and changed little after that. *Q. myrtifolia*, *Q. geminata*, and *Q. chapmanii* recovered rapidly after burning but at different rates. Cover in these three species equaled preburn values between 4 and 5 years postburn and changed little by 10 years postburn. Few changes occur through 10 years post-fire except for continued height growth. In coastal scrub that received cutting/prescribed burning treatments, cover of saw palmetto was reduced by mechanical treatment. Recovery of *Q. virginiana* was also through resprouting, which reestablished cover within 5 years postburn (Schmalzer et al. 2003). Growth rate of *Q. virginiana* was higher than shrubs in oak-saw palmetto scrub under the same treatment. The rapid growth rate of these types of scrubs has suggested that prescribed burning would need to be more frequent than often applied in these communities in order to maintain the desired structure of the vegetation. Fire return interval for oak-saw palmetto scrub has been estimated between 5 and 20 years (Schmalzer 2003). However, the length of the restoration period needs to be determined (Schmalzer et al 2003). These types of studies are also needed on the Gulf coastal scrub.

Even though some research has been conducted on the Atlantic coastal scrub, only one study has been carried out in the Gulf coastal scrub, particularly in the Panhandle. Ruth et al.

(2007) studied the effect of reintroduction of fire in long-unburned coastal scrub in Naval Live Oaks areas of the Gulf Islands National Seashore. This study addressed the effect of environmental variables and fire on plant distribution and abundance, and it found that elevation and time since fire were the most important environmental variables that affected species distribution and abundance. No study has been carried out on the west coast of Florida's peninsula. This is particularly important in CKSSR because prescribed burning has been practiced since 1985. Understanding how the scrub ecosystem in CKSSR responds to prescribed burning is critical to know the direction and rates of changes in composition and structure of scrub communities after fire and to make effective management decisions. This is the first study carried out to determine the responses of a long-unburned (since 1955) scrubby flatwoods to prescribe burning in the west coastal scrub of Florida.

Objective

Prescribed burning was applied to two long-unburned scrubby flatwoods sites in order to study the post-burn dynamics of this community. The objective was to document recovery modes and structural and compositional changes in the post-burn community. To achieve this objective, a site analysis was needed to determine if treatment and control sites were ecologically similar before burning.

Methodology

Vegetation sampling was carried out in control sites one time before prescribed burning. Sampling in treatment sites was conducted as follows: pre-burn and post-burn at 11 days, 3, 6, 9, and 12 months. I sampled the density of vegetation by placing a quadrat (4 m^2) (Figure 3-1) on specific points of the trapping grid selected by a stratified random sampling. Cover was quantified along a 2 m line that intercepted the center point of two opposite sides of each quadrat. This sampling measured the following predictor variables:

1. Ground cover of bare ground, litter, hard woody debris, and herbaceous species measured in cm.
2. Shrub cover (≤ 1.5 m tall) per species was quantified in cm.
3. Number of palmetto plants in each quadrat.
4. Number of individuals per species of trees, saplings, and seedlings, which were classified with the following criteria: oak and pine trees with dbh ≥ 7.6 cm, saplings with dbh < 7.6 cm and height ≥ 1.0 m, and seedlings with height < 1.0 m.
5. Maximum vegetation height in each quadrat.

Flowering and fruiting were not systematically surveyed, but they were recorded as encountered.

Before burning, the center of each quadrat was mapped with a Global Positioning System GPSmap 76S (Garmin) fitted with an external antenna (1-2 m accuracy). In addition, I marked the center each quadrat and the two of its north facing corners with a 20 cm wire. These wires helped to locate the quadrat after burning, and they were replaced with flags after the burn (Figure 3-1).

Counting of individual plants was done per stratum. However, there were species that could be classified as a seedling or sprout, but they could not be classified as sapling or tree because they were herbs, a lichen, a palmetto, a cactus or woody species of low height. Examples of these woody species were as follows: *Vaccinium myrsinites*, *C. ericoides*, *Osmanthus americanus*, *Gaylussacia dumosa*, *Gaylussacia nana*, *Licania michauxii*, and *Rhus copallinum*. In this case, the abundance per species was recorded without establishing an association with a particular stratum.

Counting ramets started in sites 5C and 2M at 11 and 12 days after burning, respectively (Figures 3-2, 3-3, and 3-4). Since species identification is difficult at this stage, groups of ramets were labeled and recorded with pictures. Later, these species were identified when ramets matured.

Counting of ramets was done carefully to minimize human error. Counting was carried out twice to make sure that the recorded number was accurate. If the numbers did not match, I counted slowly the third time. The two numbers that matched were selected. All ramets were counted even though they belonged to the same stem (Figure 3-5).

Recovery mode was determined by excavation of ramets. In each quadrat, I excavated 10 ramets per species in order to find seedlings or resprouting individuals at 11 days and 3 months after burning. Ramets were carefully excavated using hand tools and fingers, with care taken to preserve root systems and rhizome connections. I followed the technique used by Menge and Kohfeldt (1995) to classify seedling, resprout, or clonal ramet. Seedlings were independent plants with small root systems and no sign of pre-fire biomass (e.g. no charred stem bases). Resprouts were classified as ramets resprouting within 20 cm of pre-fire stems. Ramets more than 50 cm away from pre-fire ramets were named clonal ramets.

Univariate and multivariate data analyses were performed using SAS 9.1.3 (SAS Institute Inc. 2002-2003) and PC-ORD v.5.0 (McCune and Mefford 1999), respectively. The significance level chosen was 0.05. The data set was summarized by computing absolute and relative abundance, density, frequency (number of occupied quadrats), and mean percent cover (total distance intercepted above, below, or touching by species divided by 2 m and multiplied by 100) per species. Importance values were quantified by summing relative values for density, frequency, and mean percent cover for each species with the exception of pine trees because they did not have mean percent cover records.

A site analysis was carried out to determine if treatment and control sites at preburn conditions did not differ regarding plant species structure and composition. This analysis was critical for the experimental design because otherwise treatment effect could not be verified. This analysis started by quantifying species richness (as number of species), diversity (Simpson's index 1/D and Shannon-Wiener's index H'), and evenness (as J = H'/lnS, Pielou 1969) to measure structural and compositional differences among sites. Jaccard's and Sorensen's coefficients of similarity were also calculated to analyze the species composition among sites. After that, a cluster analysis was performed to study whether there were differences or not among sites by taking into consideration absolute abundance and mean percent cover for all herb and woody species. The dataset was standardized and outliers were deleted (McGarigal et al. 2000). In the cluster analysis, the Euclidean distance was used for the resembling matrix and Median linkage, Average linkage, and Ward minimum-variance linkage were used as fusion methods. In addition, to decide the number of significant clusters to retain, a F-ratio test in combination with Duncan's test were carried out to assess the null hypothesis that the mean for each variable was not different between multispecies clusters. Finally, a mean or median comparison was done to determine significant differences among sites for the variables found with significant differences in the F-ratio test. The idea was to determine which site(s) was (were) significantly different from other sites regarding that particular variable. Since plant species variables did not have a normal distribution according to the Shapiro-Wilk test, the data set for abundance and mean percent cover was transformed using the Arcsin function. Then, an ANOVA test and multiple comparisons (Duncan's and Bonferroni's procedures) were carried out to determine if at least two means were significantly different and which means were significantly different, respectively. If the Arcsin transformed variable did not have a normal distribution, medians were compared by using Kruskal-Wallis test and Duncan's multiple comparison procedure.

Species richness, evenness, and diversity (Simpson's and Shannon-Wiener's index) were quantified to measure and document structural and compositional changes between the pre- and the postburn community. Detrended Correspondence Analysis (DCA) was used to visualize the multivariate changes in woody species densities and mean percent cover in the preburn and postburn samples over time. Ordinations were carried out on absolute (to highlight structural changes) and relativized (to emphasize compositional changes) values of densities and mean percent cover data by using PC-ORD. Absolute values were relativized using standardization by the norm (Greig-Smith 1983). The quality of the ordinations was evaluated with the coefficient of determination that measured the proportion of the variance represented by the ordination axes. Scatter-plots between the first two axes were done to visualize structural and compositional changes. Only sampling time scores, and not species scores, were plotted for this reason. Finally, a Multi-response Permutation Procedure (MRPP) and multiple comparison was conducted using PC-ORD. MRPP is the non-parametric test analogue of MANOVA. Unfortunately, MRPP could only be performed on treatment sites pre- and 12 months postburn and control sites. MRPP can only be carried out on independent samples. Therefore, it is not the right test for a sequence of sampling through time on the same quadrats. Multi-response Block Procedure (MRBP) is a variant of MRPP, and it can be used for dependent samples. However, it requires a second matrix in PC-ORD and a balanced design (McCune and Grace 2002). The second matrix in PC-ORD is used with environmental variables that were not measured in this study, and sites had different sample sizes. Thus, MRPP was carried out to test the hypothesis of no treatment effect or no significant structural differences between treatments at 12 months postburn and treatments at preburn levels and control sites. Only the absolute density and mean percent cover of woody species were used for this analysis because the herb data set did not have a large enough sample size. Euclidean and Sorensen distances were used to calculate the distance matrices.

Results

Species List and Recovery Modes

Table 3-1 illustrates the list of species recorded in quadrats in treatment and control sites. A total of 10 herb species, 26 woody species, a lichen, and a cactus was recorded during the study. All species resprouted after burning. Even though digging was carried out on 10 individuals per species per quadrat, I did not find evidence of recovery by seeds. The only exception was *Pinus clausa* with seedlings at six months postburn in 5C. Also, the criterion of ramets more than 50 cm away from pre-fire ramets to name clonal ramets did not work. Therefore, I did not consider this recovery mode. Pre-fire ramets were burned completely in almost all quadrats.

A Site Analysis

A comparison of the structure and composition of the four sites is displayed by Table 3-2. As can be seen, control (5A and 5D) and treatment (5C and 2M) sites differed in species richness, species diversity, and evenness. In general, control sites had higher species richness, species diversity, and evenness than treatment sites. The only exception is 5D and 2M sites that had the same Shannon-Wiener's index and a similar evenness. Therefore, control and treatment sites were not ecologically similar by using these criteria.

Figure 3-6 reveals Jaccard's and Sorensen's similarity coefficients among the four study sites. All sites had a similarity higher than 54% and 63% according to Jaccard's and Sorensen's coefficients, respectively. The only exception was Jaccard's coefficient between 5C-preburn and

5D (46%). Consequently, these communities shared some structural similarities according to the coefficients.

A more powerful criterion was needed to determine if control and treatment sites were ecologically similar. Species richness, species diversity, evenness, and similarity coefficients drew different results. These criteria used the proportion of individuals and the number of species as data in a single dimension. A multivariate approach such as cluster analysis gives more insight about the actual similarity among sites.

Figures 3-7 through 3-12 show the results of the cluster analysis. Median linkage/Average linkage and Ward's minimum-variance linkage clearly displayed one cluster (Figures 3-7 through 3-10) and two clusters (Figures 3-11 and 3-12), respectively. However, looking at Ward's dendrogram, clusters were composed of a mix of sample units corresponding to treatment and control sites. As a result, I could not state that one cluster corresponds to control sites and the other to treatment sites. In order to decide the number of significant clusters to retain, Table 3-3 presents the results of the F-ratio test and Duncan's test. Out of 52 variables tested, 23 (44%) did not show results because of the small sample sizes; the means of 21 (41%) variables were not significantly different between the two clusters, and the means of eight (15%) variables were significantly different. Of these eight variables, only the mean abundances of *S. repens* and *V. myrsinites* had a significant result when means were compared among treatment and control sites (Table 3-4). The multiple comparison procedure revealed that only the median of the abundance of *V. myrsinites* in 2M preburn was significantly different from 5D and 5C preburn according to the Duncan's test. Therefore, there was enough evidence to suggest that treatment and control sites were ecologically similar, and prescribed burning effect could be determined by comparing treatment and control sites.

Postburn Recovery and Survival

Figure 3-14 shows absolute mean percent cover of bareground, litter, and debris in 5C and 2M. Bareground had postburn values higher in 2M than in 5C, but these values were not higher than 13 %. Even though the preburn value for litter was lower in 5C (83.9%) than in 2M (94.5%) and the postburn values were higher in 5C than in 2M for 3, 6, and 9 months, litter had almost exactly the same mean percent cover in both sites (5C 68.3 %; 2M 67.5%) at 12 months postburn. However, these values were lower than preburn mean percent cover in 5C, 2M, and control sites. Debris had a similar curve pattern in both sites with very low values after burning.

Preburn and postburn vegetation height in 5C and 2M is presented in Figure 3-15. As revealed by the graph, preburn height in 5C was higher than in 2M and postburn heights were similar. The vegetation took 6 months to reach one m tall, and the height remained constant until 12 months.

Tables 3-5 through 3-12 provide absolute densities, frequencies, mean percent cover, and importance values of 10 herb species in 5C and 2M. In both sites, the Family Poaceae had the highest importance value, and it was represented by several grass species. *Galactia elliottii, Solidago odora*, and *Galactia mollis* had the next three highest importance values, and *Crotalaria rotundifolia* and *Woodwardia virginica* were only recorded in 5C and 2M, respectively. The species above were counted during the preburn and/or postburn sampling periods, and the rest of the species in Tables 3-5 through 3-12 were only found during the preburn sampling period.

The postburn recovery of the most common herb species in 5C and 2M is illustrated in Figures 3-16 through 3-19. Poaceae had an upward trend for density, frequency, and importance value in 5C. Cover also had an increasing trend, but declined at nine months and increased again at 12 months. In 2M and for all variables, Poaceae leveled off from 3 to 9 months and increased one more time at 12 months. In general, Poaceae was the only taxon with preburn values and had higher values in 5C than in 2M. After burning, the behavior of the curves for *G. elliottii* was alike for all variables in both sites. *G. elliottii* increased at 3 months, decline at 6 or 9 months, and then increased again at 12 months. In general, *G. elliottii* had higher densities, frequencies, cover, and importance values in 2M than in 5C, and it surpassed density control value (5D) in 5C and both control values in 2M. *S. odora*, *G. mollis*, *C. rotundifolia*, and *W. virginica* had the same pattern for all variables in both sites. These species had low values for all variables until the 9 months and increased at 12 months, with this increase higher in 5C than in 2M. Besides *G. elliottii*, *G. mollis* was the only herb species that reached density control value (5D) at 12 months postburn in 5C.

Tables 3-5 through 3-12 display absolute densities, frequencies, mean percent cover, and importance values of 26 woody species in 5C and 2M. These tables reveal that the seven species with high values for all variables in both sites were the following: *Quercus myrtifolia*, *Serenoa repens*, *Quercus geminata*, *Lyonia ferruginea*, *Lyonia lucida*, *Quercus chapmanii*, and *Vaccinium myrsinites*. *Ilex glabra* and *Gaylussacia nana* were also common in 5C and 2M, respectively. Rare species were as follows: *Quercus nigra*, *Quercus* sp., *Ceratolia ericoides*, *Osmanthus americanus*, *Salix caroliniana*, *Smilax* sp, and *Opuntia humifusa*. *Pinus clausa*, *P. elliottii*, and *P. palustris* were present in 5C and the last two species in 2M, but few trees were recorded in quadrats. The rest of the species had moderate values in density, frequency, mean percent cover, and importance value in both sites.

The postburn absolute density recovery of the most common woody species in 5C and 2M is shown by Figure 3-20. *Q. myrtifolia* had the fastest recovery in both sites, with densities in 5C higher than in 2M. The rest of the species, *S. repens*, *Q. geminata*, *L. ferruginea*, *L. lucida*, *Q.*

chapmanii, and *V. myrsinites* had similar recovery patterns in both sites, with densities slightly higher in 2M than in 5C. However, these species did not have densities higher than 15 individuals/m^2 in both sites. *Q. myrtifolia* recuperated over preburn and control sites values. The other species achieved preburn values at 3 months, with the exception of *V. myrsinites*, and they also had equal or higher absolute densities than control sites.

Figure 3-21 reveals that the most common species had similar patterns of absolute frequency after burning in 5C and 2M. All species increased frequency at 3 months and then leveled off until 12 months, with the exception of *V. myrsinites* that continued increasing after 3 months. The only difference was that values were higher in 2M than in 5C. All species had frequencies equal or higher than preburn values at 3 months with the exception of *L. ferruginea* and *L. lucida* in 5C, and these two species and *Q. myrtifolia* and *V. myrsinites* in 2M. However, these species achieved at least a control site frequency at 3 months or later during the next 9 months.

Absolute mean percent cover for the most common species in 5C and 2M is presented in Figure 3-22. As can be seen, all species had a very similar pattern in both sites. *Q. myrtifolia* and *S. repens* had the highest cover registered in both sites. *Q. myrtifolia* increased cover at 3 months in both sites, leveled off until 9 months and grew until 12 months in 5C, and decreased at 6 months and increased until 12 months in 2M. *S. repens* had a sharp rise until 6 months, and then it tended to level out at 9 and 12 months in both sites. The other five species had an increased no higher than 5% at 3 months and remained relatively constant until 12 months, with the exception of *Q. geminata* in 5C and this species and *L. ferruginea* in 2M. These last two species in their respective sites increased frequency at 12 months with values higher than 5%. *Q. geminata*, *Q. chapmanii*, *L. ferruginea*, and *V. myrsinites*, and *G. nana* recovered preburn and/or control cover at 12 months in 2M, and *Q. myrtifolia*, *Q. geminata*, and *V. myrsinites* in 5C (Tables 3-11 and 3-7).

Figure 3-23 provides the postburn importance values of the most common species in 5C and 2M. In general, *Q. myrtifolia* had importance values higher in 5C than in 2M. It increased in its importance value at 3 months and later leveled off until 12 months in 5C. It had its highest value at 3 and 6 months, and then decreased until 12 months in 2M. *S. repens* had similar postburn recovery in both sites. It had the highest importance value at 11 and 12 days in 5C and 2M, respectively, due to their high resprout and frequency. Then, it decreased at 3 months, increased until 9 months, and declined one more time at 12 months. *Q. geminata* also had a similar recovery pattern in both sites. The importance value increased at 3 months, and then remained almost constant until 12 months. *L. ferruginea* was relatively constant in 5C, and it increased at 6 months and leveled out after that in 2M. *L. lucida* kept almost the same importance value after burning in both sites, being a little bit higher in 5C. *Q. chapmanii* resprouted at a higher level in 2M than in 5C, and the importance value at 12 days was considerably higher in 2M than in 5C for this reason. However, *Q. chapmanii*'s importance values were very similar in both sites after that with the tendency of declining. The importance value of *V. myrsinites* was relatively constant in 5C after burning, and remained steadily increasing in 2M, but at low importance values. *Q. myrtifolia*, *S. repens*, *Q. geminata*, and *Q. chapmanii* had importance values equal or higher than preburn and control sites values at 3 months in both sites. *L. ferruginea* achieved control site values during the lapse of 12 months, but not the preburn value in 5C. However, it had a higher importance value at 6 months than preburn/control sites in 2M. *L. lucida* did not reach preburn or control importance values in 5C and 2M. *V. myrsinites* recovered its preburn value at 6 months, but not control site values in 5C. It obtained control site values between 6 and 12 months, but not the preburn value in 2M.

The postburn recovery pattern described above belonged to all individuals recorded in quadrats. This included ramets, new saplings, and trees that survived fire. If I consider the calculation of the variables for ramets only, absolute density for ramets were lower than absolute density for individuals in both sites. Tables 3-13 and 3-14 show ramet absolute densities for herb and woody species in 5C and 2M and Figures 3-24 and 3-25 illustrate recovery patterns for the most common herb and woody species in 5C and 2M, respectively. As revealed by the graphs, recovery patterns for herb and woody species followed exactly the same pattern as for absolute density for individuals in both sites.

Tree mortality in both quadrats and grids is presented by Table 3-15. Tree mortality was low in quadrats at both sites (7.5%) because of the adaptation of *Quercus spp.* and *Lyonia ferruginea* to fire. These trees had 80-100% burned stem, but roots were alive. Therefore, all roots that remained alive resprouted after prescribed burning. Census on pine trees carried out in the grids before and after prescribed burning suggested that fire intensity was high enough to kill 91.6% of pine trees, including the fire adapted *P. clausa* and the fire resistant *P. palustris*.

Structural and Compositional Changes in Response to Prescribed Burning

Figure 3-26 provides preburn and postburn species richness in treatment sites in comparison with control sites. Preburn species richness was lower in 5C than in 2M. Then, species richness decreased immediately following burning in 5C and was stable in 2M. After that, species richness in 5C increased (surpassing its preburn level) and reached species richness in 5D, and obtained the highest species richness in 2M at 3 months. Later, species richness in both sites fluctuated exactly with the same pattern, being higher in 5C than in 2M. In general, species richness in 5C was higher than its preburn value and species richness in 2M after 3 months. In addition, species richness in 5C reached 5D control site value after burning, but species richness in 2M did not attain control site values.

Figures 3-27 and 3-28 indicate that species diversity and evenness, respectively, were almost constant in both sites after prescribed burning. Simpson's indexes pre- and postburn values were higher in 2M than in 5C. At both sites, Simpson's index originally decreased just after burning; then it was almost constant from several days to 9 months after burning, but increased at 12 months. Preburn species diversity was achieved in both sites at 12 months, and only 5D control value was surpassed in 2M. Shannon-Wiener's indices had the same pattern as Simpson's indices in both sites. Preburn values were reached at three months in 5C and at 12 months in 2M. Again, only 5D control index was surpassed in 2M. Evenness followed exactly the same pattern as species diversity with moderate values indicating modest predominance of the common species. Figures 3-27 and 3-28 suggest small structural and compositional changes in treatment sites until 12 months postburn.

The results of the Detrended Correspondence Analysis are shown in Table 3-16. In general, the r^2 coefficient of determination was high in both treatment sites. This indicated that the analysis provided ordinations of good quality, and a high proportion of the variance was explained by the axes. However, absolute densities had higher r^2 than relativized densities, and relativized cover had higher r^2 than absolute cover in both sites. In addition, site 5C had lower r^2 than 2M for densities and higher r^2 than 2M for cover. Except for absolute cover in 2M, the first axis had higher r^2 than the second and third axis in both sites. The scatter-plots illustrate DCA, but it is important to explain what distances represent in this ordination space.

Understanding the distance among sampling times in the ordination space is key for the interpretation of structural and compositional change of the postburn community. Each point in the species/sampling time ordination space represents the site's position on the first two axes of the ordination at a given time prior to or after fire. According to Schmalzer and Hinkle (1992a), the distance between sampling time scores (points) is an index of similarity. Sampling times that occur together are similar. Also, the distance between pre- and postburn sampling times reveal vegetation change after fire and recovery. In other words, the lengths of vectors between pre- and postburn times in ordination space indicate the vegetation change during the recovery process. Hence, the distance between sampling times (vector lengths) is an index of change and recovery (Schmalzer and Hinkle 1992a). These concepts are important to understand structural and compositional change in the scrubby flatwoods in CKSSR.

Figures 3-29 through 3-32 display absolute and relative densities and mean percent cover scores of plant species during preburn and postburn sampling times. Treatment sites 5C and 2M had structural and compositional changes through time, but they had in common the following changes: (a) a structural and compositional change after prescribed burning between preburn and three months postburn in both absolute and relativized density and cover, and (b) a structural and compositional changes between three and six months shown by the absolute and relativized cover in 5C and by absolute cover and relativized density in 2M. Treatments sites 5C and 2M differed in changes between six and 12 months. Site 5C had moderate structural and compositional changes in both absolute and relativized density and cover, but 2M had more compositional than structural changes in both variables. Now, are structural changes significant?

The results of the MRPP are shown by Table 3-17. As can be seen in this table, the test statistic T was significant at the 0.05 level for both absolute densities and mean percent cover. T described the separation among groups. The more negative is T, the stronger the separation. Therefore, at least two sites were significantly different regarding density or cover, and the multiple comparison revealed them. Absolute densities in treatment sites (5C preburn, 5C-12 months postburn, 2M preburn, and 2M-12 months postburn) and control sites (5A and 5D) were significantly different by using both Euclidean and Sorensen distances (Table 3-18). Therefore, prescribed fire did have a significant effect on absolute densities on treatment sites by changing the structure of the community between pre- and 12 months postburn. Absolute cover was only significant between control site 5D and treatment sites 5C and 2M at 12 months postburn by using Euclidean and Sorensen distances. Hence, prescribed fire did not have a significant effect on changing absolute mean percent cover between 5A and 5C/2M 12 months postburn and between preburn and 12 months postburn mean percent cover of treatment sites. This is due to the fact that preburn and control mean percent cover values were achieved by 12 months in the majority of the species (see Table 3-7 and 3-11).

Flowering and Fruiting after Prescribe Burning

Flowering and fruiting season were not synchronous in CKSSR within and among species. *V. myrsinites* started to have flowers in March and fruits in April 2006 in some sites. *G. nana* and *S. repens* started to have flowers in April, but *G. nana* had fruits in May and *S. repens* in June 2006. Other species such as *L. ferruginea*, *L. lucida*, *Ilex glabra*, and *Brevaria racemosa* started to flower in April-May and fruits in June-July 2006.

Discussion

True Control Sites

This is the first site analysis carried out to determine the veracity of control sites. In the reviewed literature about the effects of prescribed fire, control sites have been taken very loosely and without scientific rigor. In general, control sites have been assigned at random or not at random as an assumption and without scientific validation. The assumption has been made and accepted by the majority of the scientific community of fire ecology including editors of prestigious scientific journals. This study suggests that a site analysis should be the starting point for future research.

Cluster analysis in combination with ordination and univariate techniques can be used to determine if treatment and control sites are ecologically similar. Species richness, species diversity, evenness, and similarity coefficients produced contradictory results. The cluster analysis (by using three fusion techniques) in combination with discriminant analysis (to plot the first two pairs of canonical variates) and the F-ratio test (with Duncan’s and Bonferroni’s multiple comparison procedures) helped to conclude that treatment and control sites were ecologically similar. The reason for using three fusion methods relied on the purpose of conducting a site analysis: to determine if treatment and control sites were ecologically similar by carrying out a cluster analysis that really represented the structure of the data. Two space-conservative methods (Median and Average linkage) and one space-distorting method (Ward’s minimum-variance linkage) were applied to the data for this reason. According to McGarigal et al. (2000), space-conservative methods are the best choice when the objective is to reveal the true structure of the data, which is usually the case in most ecological research. Space-distorting methods do not truly represent the spatial relationship of the data because they contract or dilate the space in the immediate vicinity of the groups. A comparison between space-conservative and space-distorting methods was done to be more objective in the decision. Since, Median/Average linkage and Ward’s minimum variance linkage found one and two clusters, respectively, the F-ratio test with the multiple comparison definitively helped to determine if the structure of the dataset fit to one cluster or not. Having true control sites is critical in fire ecology research.

A site analysis must be done a priori and not a posteriori. Researchers in fire ecology need to be careful selecting control sites and test them before applying treatments to plots. Before and during this process, park managers' advice and involvement are desirable due to their experience. The success of selecting good control sites will rely on park managers' and researchers' judgment, preliminary sampling, and the statistical analysis conducted a priori.

Fire Survival and Recovery Modes

Resprouting was the main mechanism of survival and recovery with fire by the majority of the species in CKSSR. All species that occurred before burning in CKSSR exhibit resprouting life histories, except *P. clausa, P. elliottii, P. palustris*, and *C. ericoides*. Post-fire recovery through sprouting has been documented in scrubby flatwoods and sand pine scrub in Archbold Biological Station (Abrahamson 1984a, 1984b, Abrahamson and Abrahamson 1996a, 1996b), in oak-saw palmetto scrub in Kennedy Space Center (Schmalzer 2003, Schmalzer and Hinkle 1992a, 1992b, Schmalzer et al. 2003) and in sand pine scrub in National Seashore in the Panhandle (Ruth et al 2007). However, there are some species that recovery through seeding and clonal spread. For instance, *S. repens* has been reported as resprouter and seeder, *Quercus spp*. *Lyonia spp*., and *Smilax auriculata* have been cited as resprouters and clonal spreaders (Menges and Kohfeldt 1995, Menges and Hawkes 1998 see Table 3-1).

Resprouting from dormant buds, rhizome tips, root crowns, and protected meristems can account for a substantial proportion of postfire recruitment (Lyon and Stickney 1976). It is clear that almost all species (except *C. ericoides*) tolerated fire in CKSSR, and soil was a good insulator that protected underground roots and meristematic tissues as well. Tissues deeper than 5 cm rarely experience significant increase in temperature (Whelan 1995). But, did soil protect the seed bank already established during the spring 2005?

Seeds were probably not buried deep enough in the soil by the time of prescribed burning in CKSSR. Prescribed burning took place at the end of the reproductive and growing season. This was the right moment biologically and environmentally because the majority of the species already produced seeds and weather characteristics were appropriate to assure a catastrophic fire. However, fire intensity was high enough to top-kill all aboveground vegetation in treatment sites and the seed bank. No seedlings were found until 6 months postburn, and these seedlings belonged to *P. clausa*. Seeds experienced temperatures higher than 1000 °C in treatment sites, and most likely they were lying on the leaf litter and thus they were consumed. Seeds must be protected from direct heat to survive fire.

Speed of Recovery Process

Postfire changes in CKSSR were described at a very short time interval in comparison with the literature. The results obtained in CKSSR correspond to only one year postfire. Research carried out in oak-saw palmetto scrub (stands 2, 4, 8, and 24 years since burning) at Kennedy Space Center/Merritt Island National Wildlife Refuge (KSC; Schmalzer and Hinkle 1992a, 1992b, Schmalzer 2003), in long-unburned scrubby flatwoods (>35 years) at Archbold Biological Station (ABS; Abrahamson and Abrahamson 1996a), and long-unburned sand pine scrub (57 years) at Ocala National Forest (ONF; Greenberg 2003) have lasted at least 7 years. Therefore, the comparison between CKSSR and these studies was made by mainly taking into consideration the first year postburn (see Table 3-19). Even though the results obtained from CKSSR were short term, they are important because this is the first study conducted on coastal scrub in the west coast of the Florida Peninsula.

Another factor to take into consideration is how variables were measured among studies. Mean height, species richness, and mean percent cover of bareground and species presented in Table 3-19 were the only variables and the most common species shared among these three studies. Mean percent cover in CKSSR was quantified for herb species and for shrub species in the stratum ≤ 1.5 m tall and the results are comparable with ABS and ONF because these studies did not establish any stratum. At KSC, Schmalzer (2003) measured cover at two stratum: <0.5 m and >0.5 m. Therefore, the data from these two strata were summed to make them comparable with the other three studies. Unfortunately, other studies carried out in ABS, KSC, and Naval Live Oaks/Gulf Island National Seashore (NLO; Ruth et al. 2007) could not be included in the comparison because they did not quantify the same variables through time. However, important results from these studies are cited to highlight particular points in this discussion.

Bareground recovery is slow through time in long-unburned scrubs. Since, long-unburned scrubs have zero or very low mean percent of bareground cover, fire considerably increases bareground cover during the first 6 to 12 months postburn, then it tends to decrease as *Quercus* spp., *Lyonia* spp., and *S. repens* increase density and cover through time. In KSC, bareground increased significantly after burning until 6 months (22.9%), but declined rapidly to 0.7% at 36 months. In ONF, bareground increased significantly after burning, reaching its maximum at 16 months postburn (25%), and by 101 months postburn (8%) still was not near preburn level (0.00%). Even though the study in CKSSR was at short term, the recovery of the vegetation was so fast that bareground had 3.8% and 6.8% at 6 and 12 months postburn, respectively. Probably, bareground in CKSSR will recuperate to the preburn level faster than in the other two study areas.

The recovery process for litter and debris has not been well documented in the literature. Greenberg (2003) reported these values in a sand pine stand in ONF. Litter was recorded as depth of litter layer. It decreased from preburn level of 6.5 cm to 2.0 cm and 2.5 cm at 5 and 16 months postburn, respectively, and then slowly decreased to 1% at 101 months postburn. Debris had a preburn value of 0.2%, which was constant until 16 months, then it increased to 8.4% at

101 months. In CKSSR, litter and debris recovered faster than in ONF, achieving litter more than 70% and debris more than 30% of the preburn level in treatment sites in 12 months.

Vegetation height was one of the variables with the slowest recovery process. In KSC, mean pre-burn height was 1.08 m and reached 0.32 m and 0.50 m at 6 and 12 months, respectively. Height growth continued throughout the postburn period reaching preburn value at 85 months. In ONF, the mean height of the vegetation was determined by measuring the height of *Q. myrtifolia* and *Sabal etonia*. Table 3-19 only shows the mean height of *S. etonia* because it was the highest data recorded at 5 and 16 months. By 5 months postburn, *S. etonia* reached its preburn value (1 m), and *Q. myrtifolia* needed almost 64 months to achieve its preburn mean value of 1.25 m. In CKSSR, the mean height of the vegetation recovered slower than in the other two study areas. It was 31% of the preburn value at 12 months, while in KSC was 46% of the preburn value.

The recovery process in KSC is described below. *Aristida stricta* was the most common herb species and recovered preburn value in six months. Other herb species also found in CKSSR such as *Carphephorus* spp. and *Galactia elliottii* were not recorded during the preburn sample, and they were censused with low cover during the postburn period. Regarding woody species in general, *Q. myrtifolia*, *S. repens*, and *L. lucida* were the dominant species (Table 3-19), with *Q. geminata* and *Q. chapmanii* also relatively common. At the <0.5 m stratum, *Q. myrtifolia* and *L. lucida* increased cover more than 5 times the preburn levels after burning. Then, cover decreased at four years for *Q. myrtifolia* and two years for *L. lucida*, and reached preburn level after seven and five years, respectively. At the >0.5 m stratum, *Q. myrtifolia* and *L. lucida* significantly decreased cover after one year postburn and needed five years to recover pre-burn values. *S. repens* did not have a high increase in cover in the <0.5 m stratum after burning, and it had a low cover relatively constant through time. *S. repens* reestablishes cover faster than woody shrubs at >0.5 m stratum. It increased cover right after prescribed burning and reached the preburn level between one and one and a half year. *Q. geminata* and *Q. chapmanii* had a low increase in cover after burning at the <0.05 m stratum. Then, they gradually reduced cover after three years. Both species decreased cover after burning at the >0.5 m stratum and obtained preburn levels after 5 years postburn. *L. lucida* recovered cover between 4 and 5 years. Schmalzer and Hinkle reported shifts in dominance after fire due to differences in recovery rates in shrub species and *S. repens. V. myrsinites* had a low increase after burning that persisted relatively constant through time at the <0.05 m stratum, and it started to appear after three years at the >0.05 m stratum, and it kept low cover after that.

In ABS, Abrahamson and Abrahamson did not show data of cover for herb species during five years of postburn period. They only presented the average percent cover for five 200-m transects of seven species not found in CKSSR. However, they reported that long-unburned scrubby flatwoods contained fewer herb species during the postburn period than recently burned scrubby flatwoods. A possible explanation suggested by the authors was that fire had the effect of creating or enlarging gaps making the persistence of herbs (gap specialists) sensitive to time since fire. Long fire-free periods may allow shrubs to clonally spread into gaps and make the gap-specialist herbs disappear through time. *S. repens*, *Q. chapmanii*, and *Q. geminata* were dominant species. Cover of *Q. chapmanii*, *Q. geminata*, *Q. minima*, *Lyonia fruticosa*, *L. lucida*, *V. myrsinites* and *M. cerifera* returned to or exceeded preburn levels within three years following fire. *S. repens* reached preburn level in two years, did not maintain preburn dominance, but cover was higher than 15% through time.

In ONF, *Rynchospora megalocarpa* was the dominant herb species and recuperated its preburn value (1.84%) in 16 months. Species common to CKSSR were: *Clitoria mariana* (0.01-0.80%), *G. elliottii* (0.02%), and *Zamia pumila* (0.02-0.53%), and they occurred with low mean percent cover. Of 28 herb species, 19 were absent from transects prior to the burn and occurred on transects during postburn samples with low cover (0.01-1.53%). In general, scrub woody species composition and cover were similar to preburn values after 16 months. *Q. myrtifolia* was the most dominant shrub before fire, recovered rapidly, and attained 67% of preburn cover level at 16 months postburn. *Q. geminata* regained 84% of its preburn level in 16 months. Mean percent cover of *Q. chapmanii* was 0.98% preburn, and it almost achieved this value at 16 months (0.93%). *S. repens* recovered 75% of its preburn level and *L. ferruginea* surpassed its preburn level in 16 months. In general, recovery rates among species did not result in long-term shifts in species dominance because *Sabal etonia* recovered its preburn level faster than *Q. myrtifolia*, but *Q. myrtifolia* continued to be the dominant species.

Comparing the recovery pattern in CKSSR with KSC, ABS, and ONF, we found a different story for herb and woody species. *Galactia elliottii* was the most common herb species in CKSSR, but with limited comparison because it was only recorded by few sampling periods in KSC and ONF. *Carphephorus corymbosus* was only recorded in control site 5D at CKSSR, and it had low cover in almost all samples in KSC. *Clitoria mariana* and *Zamia pumila* were sampled only in 5A and 5D, respectively, in CKSSR, and they also had low cover in almost all samples in ONF. At CKSSR, *G. elliottii* and *G. moilis* were the only herb species sampled on control sites and during the postburn period, while *Crotalaria rotundifolia*, *Solidago odora* and *Woodwardia virginica* were registered only during the postburn period. Therefore, few species are present with low cover before and after burning, and other species colonize gaps available after burning for a temporal use (with low cover) and until they are replaced by the dominant and growing woody vegetation.

In general, the scrub woody vegetation in KSC, ABS, ONF, and CKSSR returned to preburn conditions rapidly after a high-intensity prescribed burn. This aspect has been reported by other studies in scrub vegetation in Florida (Abrahamson 1984a, 1984b; Menges et. al. 1993; Schmalzer et al. 2003; Ruth et al. 2007). As can be seen in Table 3-19, CKSSR and ONF had the fastest recovery. However, I did not investigate cover after 12 months in CKSSR. The recovery time for the other studies suggest that there is variation depending on the type of scrub. Regarding the dominant species, *Q. myrtifolia* and *S. repens* were the most common species in at least three study areas. There was a shift in dominance after fire for shrubs and palmetto in KSC, for *S. repens* in ABS, and for *Q. chapmanii* and *L. ferruginea* in CKSSR (see Figure 3-22).

Although *P. clausa* is a species adapted to fire, its seedlings take some time to appear after fire. In ABS, *P. clausa* appearance is delayed three years postburn (Abrahamson 1984b). In ONF, P. clausa seedlings were established after 5 months postburn (Greenberg 2003). In NLO (Ruth et al. 2007), *P. clausa* seedlings were absent eight months to two years after burning from sand pine scrub. In CKSSR, seedlings appeared six months postburn. Hence, *P. clausa* seedlings appear after several months to several years in the scrub and very little is known about seedlings survival and establishment.

Even though several vegetation variables returned to preburn conditions in CKSSR during an interval of 12 months, this time was still too short to predict that treatment sites would be restored to scrubby flatwoods without a history of fire suppression. According to Abrahamson and Abrahamson (1996a, 1996b) and Baker (1992, 1994), prescribed burning does not necessarily reestablish the preburn conditions in all landscapes after several years of fire suppression. This is an aspect that needs to be determined in Cedar Key with continued monitoring of the study sites. Schmalzer and Boyle (1998), Schmalzer and Adrian (2001), and Schmalzer et al. (2003) recommended a combination of mechanical treatment and prescribed burning in restoring long-unburned scrub vegetation. Mechanical cutting should be used only one time to reestablish shrub vegetation structure, which can be maintained with periodic prescribed burning after that. All mechanical treatments result in some loss of *S. repens* cover, and this loss persists (Schmalzer et al. 2003). So, mechanical treatment must be applied carefully.

Whether the reintroduction of fire to long-unburned scrub might produce or not the desired effect of returning the association to a state similar to scrubs without fire suppression will rely on fire intensity and the season of burning. The reintroduction of frequent and low-intensity fires to sand pine scrub may shift scrub to sandhill (Myers 1985). In contrast, a single, high-intensity fire in sand pine scrub may make possible the perseverance of the scrub stand (Myers 1985, Menges et al. 1993, Menges and Hawkes 1998). Instead, a single-low intensity fire might facilitate the shift of scrub toward xeric hammock. This situation would occur if fire increases the abundance of the sprouter species, and consequently repress the regeneration of obligate seeders such as sand pine and herb species. According to Abrahamson & Abrahamson (1996b), a single fire may not be effective at restoring long-unburned scrubby flatwoods to states characteristic of more recently burned stands. Several fires may be needed before scrubby flatwoods are returned to communities similar to those without fire suppression.

Community Shift in Response to Prescribed Burning

The scrubby flatwoods community in CKSSR had structural and community changes after prescribed burning, and they were reported for an interval of 12 months. As previously mentioned, studies conducted in KSC, ABS, and ONF have datasets for at least 7 years, which constraint the comparison with CKSSR. This study presents a comparison between CKSSR and these three studies regarding species richness, species diversity (Shannon-Wiener's index), evenness (H' / ln S), and the results of the Detrended Correspondence Analysis mainly restricted to 12 months postburn.

Mean species richness in KSC increased in the <0.5 m stratum, but declined in the >0.5 m stratum, during the first 12 months. Considering both strata together, there was an increase in the species richness 12 months postburn. Species diversity and evenness were not measured in this study. The DCA carried out only with mean percent cover indicated that the composition of the scrub varied along a gradient closely related to the depth of the water table with oak dominating drier sites and *S. repens* and *I. glabra* in wetter places. The species ordination located oak species to the left and *S. repens, I. glabra*, and *M. cerifera* to the right of the axis. This pattern was found in all oak-dominated transects and saw palmetto-dominated transects. The other transects with a mixed oak-saw palmetto composition before burning located sample units in the middle of the ordination axis. Both, oak-dominated and saw palmetto-dominated transects in the ordination space, returned to preburn locations after three years. Vector lengths were greater for the mixed oak-saw palmetto transects than those dominated by *S. repens*, indicating a greater degree of change in the mixed oak-saw palmetto transects.

In one long-unburned (>35 yr) and two recently burned (<20 yr) scrubby flatwoods in ABS, species richness was constant in long-unburned and decreased in recently burned during the first 12 months. Later, species richness increased in all stands relative to preburn levels until 36 months, and increases were most pronounced at recently burned scrubby flatwoods. Also, species richness of long-unburned scrubby flatwoods was reduced relative to preburn levels after 36 months. Species diversity increased and reached the average preburn index in the long-unburned stand at 12 months postburn. However, species diversity increased just after burning and then decreased in recently burned stands in the first 12 months. But the index at 12 months was higher than preburn index. After the first year, species diversity increased and remained high for the recently burned stands, which had a higher index through time than the long-unburned stand. Species diversity in the long-unburned stand decreased at 24 months and 48 months postburn and then linearly increased through time after that. Overall, evenness was reduced after fire. Long-unburned stands achieved preburn evenness at approximately 72 months postburn; however, recently burned stands did not. The DCA was carried out to visualize the multivariate changes in species dominance (percentage of cover based on crown intercepts) in the preburn and postfire samples. The analyses, based on absolute and relativized dominance, showed that scrubby flatwoods were very stable following fire. Changes related to composition were more noticeable than structural changes. In general, the effect of fire on these stands caused slight changes in stand structure and some degree of shifts in stand composition regardless of time since fire. These results are similar to the conclusions found from other studies in Florida (Abrahamson 1984a, 1984b; Abrahamson et al. 1984; Abrahamson & Hartnett 1990; Schmalzer & Hinkle 1992a).

In a sand pine scrub stand in ONF, herbaceous species richness increased within 5 months postburn, peaked at 16 months, and declined by 40 months postburn. Woody species richness decreased immediately after fire (five weeks), then increased until 28 months and remained constant after that. In general, species richness increased during the first 12 months. The increase followed by the gradual decline of herbaceous species richness appeared to be related with gradual increases in shrub cover and decreases in bare ground availability. Carrington and Keeley (1999) suggested that the low postburn seedling recruitment in sand pine scrub is probably due to the elimination of suitable microsites by resprouting shrubs. Greenberg (2003) did not quantify species diversity, evenness, and did not carry out a DCA.

The results found in CKSSR are similar to the results found in the other studies. KSC, ONF, and CKSSR had an increased of species richness, but ABS did not during the first 12 months. This increase in species richness is expected due to the disturbance caused by fire, the amount of species resprouting and clonal spread, and new herb species that are temporarily colonizing gaps. Species diversity in the long-unburned scrubby flatwoods in ABS and in CKSSR achieved preburn levels in 12 months postburn. Evenness regained preburn levels in ABS and CKSSR, but was faster in CKSSR. The information obtained from species richness, species diversity, and evenness suggests little structural and compositional changes at short term in CKSSR and at long term in ABS. In contrast, DCA was able to reveal results not detected by these indices.

The DCA revealed both structural and compositional changes at short term in CKSSR and more compositional than structural changes at long term in ABS. In CKSSR, there were both structural and compositional changes with similar magnitude during the first 3 months according to the lengths of the vectors (Figure 3-29 through 3-32). These changes were expected because many species were resprouting simultaneously. Between 3 and 12 months, there were also structural and compositional changes because almost all species already resprouted (little variation in species richness and diversity), their frequencies were almost constant thorough time (Figure 3-21), and mainly the density and cover of *Q. myrtifolia* and the cover of *S. repens*, *Q. geminata*, and *L. ferruginea* increased through time (Figures 3-20 and 3-22). In a low-intensity fire on long-unburned sand pine scrub in ABS, Abrahamson and Abrahamson (1996b) reported that the largest amount of structural and compositional change occurred immediately after prescribed fire, just between the preburn and 1-year postburn censuses. In contrast, changes between the 1-year and 2-year censuses after fire were primarily compositional; little structural change was measured.

The DCA carried out in CKSSR on mean percent cover also showed that the composition of the scrub varied along a gradient associated with the water table. Figures 3-33 and 3-34 presents the ordination found in KSC and in CKSSR with preburn samples of mean percent cover, respectively. Comparing both figures, we can see that oaks are in the left side of the ordination (drier places) and *S. repens* and *I. glabra* are in the right side of the ordination (wetter places). In Cedar Key, *M. cerifera*, *L. lucida*, and *Q. geminata* are almost in the middle of the gradient, but not in KSC. However, the position of *L. lucida* is not exactly to the left and the location of *M. cerifera* is not exactly to the right in the ordination space in KSC.

Age at First Flowering after Prescribed Fire

Age at first flowering might vary among species within a single community, between populations of a single species, and within one population. In addition, year of first flowering for several species might vary among sites in the same study area (Whelan 1995). In CKSSR, flowering was not synchronous within and among species and among stands without burning. Flowering started in March 2006 after the prescribed burnings in April and May 2005. Probably, sprouting species delayed flowering after prescribed fire because the energy produced through photosynthesis was devoted to vegetative growth. In sand pine scrub in ABS (Abrahamson and Abrahamson 1996b), *S. etonia* did not flower until spring the year following a February burn. In contrast, in ONF (Greenberg 2003), *S. etonia* flowered within 5 weeks and fruited within 5 months. The factors that affect post-fire sprouting are likely to affect post-fire flowering when the number of flowers or inflorescences per plant are determined by the number of active shoots. Each shoot sprouting after fire can produce a terminal inflorescence (Whelan 1995). Therefore, the season of burn, fire intensity, or both may be important factors that affect flowering in sprouting species. Unfortunately, the effects of these factors on flowering of sprouting species have rarely been quantified.

Table 3-1. List of plant species recorded in quadrats in treatment and control sites in Cedar Key Scrub State Reserve.

Species	Common Name	Family	Recovery Mode
Herbaceous			
Agalinis filifolia	Seminole False Foxglove	Orobanchaceae	Resprouter (1)
Asclepias sp	Milkweed	Apocynaceae	Resprouter (2)
Carphephorus corymbosus	Florida Paintbrush	Asteraceae	Resprouter (2)
Clitoria mariana	Atlantic Pigeonwings	Fabaceae	Resprouter (3)
Crotalaria rotundifolia	Rabbitbells	Fabaceae	Resprouter
Galactia elliottii	Elliottis Milkpea	Fabaceae	Resprouter (2)
Galactia mollis	Soft Milkpea	Fabaceae	Resprouter
Solidago odora	Chapman's Golden Rod	Asteraceae	Resprouter (*) (2)
Woodwardia virginica	Virginia Chain Fern	Blechnaceae	Resprouter (1)
Zamia pumila	Florida Arrowroot	Zamiaceae	Resprouter (3)
Woody			
Pinus clausa	Sand Pine	Pinaceae	Obligate seeder (2)
Pinus elliottii	Slash Pine	Pinaceae	Obligate seeder (♦) (2)
Pinus palustris	Long-leaf Pine	Pinaceae	Obligate seeder (♦)
Quercus geminata	Sand Live Oak	Fagaceae	Resprouter (♠) (2)
Quercus myrtifolia	Myrtle Oak	Fagaceae	Resprouter (♠) (2)
Quercus chapmanii	Chapman Oak	Fagaceae	Resprouter (♠) (2)
Quercus minina	Runner Oak	Fagaceae	Resprouter (♠) (2)
Quercus nigra	Water Oak	Fagaceae	Resprouter
Quercus sp		Fagaceae	Resprouter
Brevaria racemosa	Tar Flower	Ericaceae	Resprouter (2)
Ceratolia ericoides	False Rosemary	Empetraceae	Obligate seeder (2)
Ilex glabra	Gallberry	Aquifoliaceae	Resprouter
Lyonia ferruginea	Rusty Lyonia	Ericaceae	Resprouter (2)
Lyonia fruticosa	Stagger-bush	Ericaceae	Resprouter (♠) (2)
Lyonia lucida	Fetterbush	Ericaceae	Resprouter (♠) (2)
Myrica cerifera	Wax Myrtle	Myricaceae	Resprouter (♠) (2)
Gaylussacia dumosa	Dwarfhuckleberry	Ericaceae	Resprouter (♠) (2)
Gaylussacia nana	Dangleberry	Ericaceae	Resprouter (1)
Licania michauxii	Gopher Apple	Chrysobalanaceae	Resprouter (♠) (2)
Osmanthus americanus	Wild Olive	Oleaceae	Resprouter
Rhus copallinum	Winged Sumae	Anacardiaceae	Resprouter (2)
Salix caroliniana	Carolina Willow	Salicaceae	Resprouter
Serenoa repens	Saw Palmetto	Arecaceae	Resprouter-seeder
Smilax auriculata	Catbrier	Smilacaceae	Resprouter (*) (2)
Smilax sp		Smilacaceae	Resprouter
Vaccinium myrsinites	Blueberry	Ericaceae	Resprouter (♠) (2)
Cladonia evansii	Deer Moss	Cladoniaceae	Seeder (2)
Opuntia humifusa	Devil's tongue	Cactaceae	Resprouter

Codes: (*) Resprouter, clonal spreader, and seeder. (♠) Resprouter and clonal spreaders. (♦) Obligate seeder and survivor. (1) United States Department of Agriculture website. (2) Menge & Kohfeldt (1995). (3) Greenberg (2003).

Table 3-2. Species richness, Simpson's index , Shannon-Wiener's index , and Shannon-Wiener's evenness for preburn conditions in control (5A & 5D) and treatment (5C & 2M) sites in Cedar Key Scrub State Reserve.

Site	Richness	Simpson	Shannon-Wiener	Evenness
5A	26	10.64	2.54	0.6981
5D	24	7.06	2.28	0.6271
5C	17	5.03	1.99	0.5483
2M	20	8.41	2.28	0.6265

Table 3-3. Multiple mean comparison (Duncan's test) between clusters created by using Euclidean distances and Ward's minimum variance linkage fusion. Abundance and mean percent cover were standardized. Significant level = 0.05.

	Cluster 1		Cluster 2			
Variable	n	Mean	n	Mean	F value	P-value
Galactia elliottii (abundance)	34	-0.2873	47	-0.2635	0.08	0.7742
Galactia elliottii (cover)	34	-0.2375	47	-0.2089	0.13	0.7234
Quercus geminata (abundance)	34	-0.1107	47	-0.1272	0.01	0.9163
Quercus geminata (cover)	34	-0.1750	47	-0.2104	0.14	0.7130
Quercus myrtifolia (abundance)	34	-0.6000	47	0.6634	41.47	<.0001
Quercus myrtifolia (cover)	34	-0.5288	47	0.7118	38.54	<.0001
Quercus chapmanii (abundance)	34	-0.2150	47	-0.1768	0.08	0.7792
Quercus chapmanii (cover)	34	-0.1741	47	-0.2252	0.42	0.5173
Quercus minima (abundance)	34	-0.0354	47	-0.1105	0.25	0.6219
Quercus minima (cover)	34	-0.2316	47	-0.1840	0.72	0.3984
Quercus sp (abundance)	34	-0.1589	47	-0.1404	0.03	0.8578
Brevaria racemosa (abundance)	34	-0.0374	47	-0.1302	2.86	0.0945
Ceratolia ericoides (cover)	34	-0.2030	47	-0.1770	0.72	0.3984
Ilex glabra (abundance)	34	-0.0432	47	-0.2026	5.19	0.0254
Lyonia ferruginea (abundance)	34	-0.2414	47	-0.0951	1.13	0.2919
Lyonia ferruginea (cover)	34	-0.4303	47	-0.1164	6.85	0.0106
Lyonia fruticosa (abundance)	34	-0.1792	47	-0.2084	1.45	0.2315
Lyonia lucida (abundance)	34	0.1317	47	-0.1755	3.74	0.0567
Lyonia lucida (cover)	34	0.2238	47	-0.3472	15.7	0.0002
Myrica cerifera (abundance)	34	-0.2175	47	-0.1783	0.11	0.7398
Myrica cerifera (cover)	34	-0.1522	47	-0.1760	0.26	0.6135
Gaylussacia dumosa (abundance)	34	-0.1296	47	-0.1108	0.07	0.7939
Gaylussacia nana (abundance)	34	-0.1939	47	-0.2177	0.18	0.6720
Licania michauxii (abundance)	34	-0.0984	47	-0.0978	0.00	0.9909
Serenoa repens (abundance)	34	0.6573	47	-0.3788	35.23	<.0001
Serenoa repens (cover)	34	1.0759	47	-0.4110	72.16	<.0001
Smilax auriculata (abundance)	34	-0.1325	47	-0.1809	1.39	0.2421
Vaccinium myrsinites (abundance)	34	-0.3902	47	-0.1371	5.21	0.0251
Vaccinium myrsinites (cover)	34	-0.2928	47	-0.1172	2.52	0.1165

Table 3-4. T test, ANOVA test, and Kruskal-Wallis test for comparing means and medians among treatment and control sites under preburn conditions in Cedar Key Scrub State Reserve. Data for T test and ANOVA were standardized. Test for ANOVA and Kruskal-Wallis is F test and Chi-squared test, respectively. Significant level = 0.05.

Test	Species	5A	5D	5C pre	2M pre	Test	DF	P-value
T	*Ilex glabra* (*)	-0.2907		0.1642		-1.1570	16	0.2642
ANOVA	*Quercus myrtifolia* (*)	0.0721	0.0655	-0.1537	-0.1519	0.7000	3	0.5523
	Quercus myrtifolia (%)	-0.0683	0.1724	0.0835	0.0867	0.3300	3	0.8014
	Lyonia ferruginea (%)	0.2617	0.2827	0.4946	0.3457	0.1700	3	0.9169
	Lyonia lucida (%)	0.0167	0.1522	-0.0267	-0.6480	1.7500	3	0.1654
	Serenoa repens (%)	-0.2689	0.1440	0.0508	0.0926	0.8600	3	0.4657
K-W	*Serenoa repens* (*)	4.0000	3.0000	4.0000	3.0000	7.9563	3	0.0469
	Vaccinium myrsinites (*)	5.0000	3.0000	2.0000	10.0000	18.7388	3	0.0003

Codes: (*) mean abundance. (%) mean percent cover.

Table 3-5. Pre- and postburn absolute densities for all species in 5C in Cedar Key Scrub State Reserve. Absolute densities for control sites 5A and 5D are also shown. d = days. M = months.

	Control			Time after burning				
Species	5A	5D	Pre	11 d	3 M	6 M	9 M	12 M
Herbaceous								
Poaceae	0.37	0.14	0.03	0.09	0.61	1.15	0.98	1.90
Solidago odora	0.00	0.00	0.00	0.00	0.12	0.11	0.06	1.82
Galactia elliottii	2.46	1.09	0.00	0.00	2.66	1.26	0.00	1.35
Galactia mollis	0.00	0.18	0.00	0.00	0.04	0.05	0.00	1.22
Crotalaria rotundifolia	0.00	0.00	0.00	0.00	0.08	0.07	0.07	0.03
Agalinis filifolia	0.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00
Asclepias sp	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Carphephorus corymbosus	0.00	0.19	0.00	0.00	0.00	0.00	0.00	0.00
Clitoria mariana	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Woodwardia virginica	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Zamia pumila	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00
Woody								
Quercus myrtifolia	10.91	12.17	12.53	1.46	45.58	43.46	47.76	42.11
Lyonia ferruginea	2.09	5.06	3.87	0.58	8.32	10.39	11.22	11.60
Quercus geminata	2.90	2.24	2.20	0.38	13.20	12.68	13.24	11.58
Lyonia lucida	7.03	10.70	4.49	0.55	8.23	8.83	8.87	8.88
Quercus chapmanii	3.01	2.01	2.79	0.29	9.99	8.58	7.33	6.43
Serenoa repens	1.14	1.01	1.41	1.80	3.31	4.32	4.34	4.94
Ilex glabra	2.98	0.55	0.13	0.06	4.63	4.91	4.74	3.55
Licania michauxii	0.76	0.20	0.00	0.00	2.65	2.43	0.90	2.34
Lyonia fruticosa	1.88	0.00	0.25	0.15	1.24	1.85	2.22	1.71
Vaccinium myrsinites	5.08	2.04	0.51	0.00	0.97	1.43	1.44	1.69
Gaylussacia dumosa	5.07	1.37	0.00	0.00	1.69	1.41	1.46	1.40
Gaylussacia nana	0.31	5.30	0.03	0.00	1.32	1.19	0.28	1.13
Brevaria racemosa	0.57	0.05	0.00	0.03	1.92	2.00	1.65	1.06
Myrica cerifera	1.04	0.47	0.85	0.01	0.43	0.77	0.69	0.50
Rhus copallinum	0.00	0.00	0.00	0.00	0.25	0.28	0.00	0.31
Pinus clausa	0.00	0.00	0.02	0.01	0.01	0.03	0.08	0.25
Smilax auriculata	0.12	0.08	0.00	0.00	0.03	0.11	0.17	0.12
Quercus minina	2.50	1.70	1.21	0.00	0.30	0.38	0.35	0.09
Pinus palustris	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.08
Pinus elliottii	0.04	0.00	0.04	0.00	0.00	0.01	0.02	0.05
Quercus nigra	0.00	0.00	0.08	0.00	0.00	0.00	0.00	0.00
Quercus sp	0.07	0.17	0.11	0.00	0.02	0.00	0.00	0.00
Ceratolia ericoides	0.03	0.02	0.00	0.00	0.00	0.00	0.00	0.00
Osmanthus americanus	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00
Salix caroliniana	0.00	0.00	0.00	0.00	0.03	0.02	0.00	0.00
Smilax spp	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cladonia evansii	3.51	1.80	2.26	0.00	0.00	0.00	0.00	0.00
Opuntia humifusa	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 3-6. Pre- and postburn absolute frequencies for all species in 5C in Cedar Key Scrub State Reserve. Absolute frequencies for control sites 5A and 5D are also shown. d = days. M = months.

	Control			Time after burning				
Species	5A	5D	Pre	11 d	3 M	6 M	9 M	12 M
Herbaceous								
Poaceae	12	6	4	3	15	17	18	25
Galactia elliottii	31	29	0	0	35	28	1	20
Galactia mollis	0	1	0	0	1	2	0	11
Solidago odora	0	0	0	0	5	5	4	8
Crotalaria rotundifolia	0	0	0	0	1	1	1	1
Agalinis filifolia	0	1	0	0	0	0	0	0
Asclepias sp	1	0	0	0	0	0	0	0
Carphephorus corymbosus	0	1	0	0	0	0	0	0
Clitoria mariana	2	0	0	0	0	0	0	0
Woodwardia virginica	0	0	0	0	0	0	0	0
Zamia pumila	0	1	0	0	0	0	0	0
Woody								
Quercus myrtifolia	34	44	35	29	36	35	35	35
Quercus geminata	29	34	30	14	34	33	34	34
Serenoa repens	27	29	33	31	35	35	35	34
Quercus chapmanii	32	35	27	12	29	30	31	31
Lyonia ferruginea	19	34	27	18	20	24	24	26
Lyonia lucida	28	38	31	13	28	25	24	24
Vaccinium myrsinites	34	27	14	0	14	20	19	23
Quercus minina	26	40	15	0	9	11	18	15
Pinus clausa	0	0	2	1	1	2	5	14
Myrica cerifera	14	12	15	2	8	14	13	13
Licania michauxii	5	6	0	0	13	15	5	13
Ilex glabra	13	4	3	1	6	11	9	9
Rhus copallinum	0	0	0	0	11	11	0	9
Smilax auriculata	5	4	0	0	1	7	8	7
Gaylussacia nana	1	29	2	0	8	7	5	6
Pinus elliottii	2	0	3	0	0	1	2	4
Lyonia fruticosa	13	0	4	3	3	3	3	4
Gaylussacia dumosa	24	22	0	0	6	3	3	4
Brevaria racemosa	8	2	0	2	2	3	3	3
Pinus palustris	3	0	0	0	0	0	0	2
Quercus nigra	0	0	1	0	0	0	0	0
Quercus sp	3	12	3	0	1	0	0	0
Ceratolia ericoides	3	6	0	0	0	0	0	0
Osmanthus americanus	0	1	0	0	0	0	0	0
Salix caroliniana	0	0	0	0	1	1	0	0
Smilax spp	1	0	0	0	0	0	0	0
Cladonia evansii	8	12	14	0	0	0	0	0
Opuntia humifusa	1	0	0	0	0	0	0	0

Table 3-7. Pre- and postburn absolute mean % cover of herb and woody species in 5C in Cedar Key Scrub State Reserve. Absolute mean percent cover for control sites 5A and 5D are also shown. d = days. M = Months.

	Control		Time after burning					
Species	5A	5D	Pre	11 d	3 M	6 M	9 M	12 M
Herbaceous								
Poaceae	0.47	0.00	0.14	0.04	0.46	2.41	1.92	2.07
Galactia elliottii	2.06	1.76	0.00	0.00	1.83	0.13	0.00	0.51
Solidago odora	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.47
Galactia mollis	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.11
Agalinis filifolia	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Asclepias sp	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Carphephorus corymbosus	0.00	0.26	0.00	0.00	0.00	0.00	0.00	0.00
Clitoria mariana	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Crotalaria rotundifolia	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Woodwardia virginica	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Zamia pumila	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.00
Woody								
Quercus myrtifolia	20.07	27.36	28.43	0.00	15.21	16.83	16.07	24.95
Serenoa repens	23.95	24.39	28.66	0.00	13.26	19.32	22.48	23.00
Quercus geminata	6.75	0.64	2.71	0.00	4.26	4.51	3.76	7.64
Lyonia lucida	8.94	18.00	4.35	0.00	2.05	2.35	2.82	3.58
Lyonia ferruginea	4.22	5.70	5.32	0.00	1.44	1.84	3.28	3.10
Quercus chapmanii	2.89	5.97	2.75	0.00	2.12	2.47	1.75	2.61
Ilex glabra	2.28	0.00	0.28	0.00	0.57	2.18	0.77	1.27
Myrica cerifera	0.39	1.32	1.87	0.00	0.42	0.43	0.95	0.90
Lyonia fruticosa	1.48	0.00	0.07	0.00	0.00	0.55	0.42	0.73
Vaccinium myrsinites	3.55	2.80	0.37	0.00	0.05	0.34	0.24	0.59
Rhus copallinum	0.00	0.00	0.00	0.00	0.16	0.31	0.00	0.26
Licania michauxii	0.04	0.00	0.00	0.00	0.25	0.08	0.00	0.14
Gaylussacia dumosa	0.43	0.06	0.00	0.00	0.09	0.07	0.00	0.10
Gaylussacia nana	0.28	1.60	0.00	0.00	0.15	0.00	0.00	0.07
Pinus clausa	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinus elliottii	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinus palustris	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Quercus minina	0.59	0.27	0.23	0.00	0.00	0.00	0.00	0.00
Quercus nigra	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Quercus sp	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Brevaria racemosa	1.44	0.72	0.00	0.00	0.00	0.00	0.00	0.00
Ceratolia ericoides	2.71	7.49	0.00	0.00	0.00	0.00	0.00	0.00
Osmanthus americanus	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Salix caroliniana	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Smilax auriculata	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Smilax spp	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cladonia evansii	1.05	0.00	0.33	0.00	0.00	0.00	0.00	0.00
Opuntia humifusa	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 3-8. Pre- and postburn absolute importance values of herb and woody species in 5C in Cedar Key Scrub State Reserve. Absolute importance values for control sites 5A and 5D are also displayed. d = days. M = Months.

	Control			Time after burning				
Species	5A	5D	Pre	11 d	3 M	6 M	9 M	12 M
Herbaceous								
Poaceae	0.0418	0.0164	0.0173	0.0769	0.0619	0.1031	0.1027	0.1120
Galactia elliottii	0.1457	0.1055	0.0000	0.0000	0.1736	0.0932	0.0032	0.0722
Solidago odora	0.0000	0.0000	0.0000	0.0000	0.0162	0.0243	0.0135	0.0446
Galactia mollis	0.0000	0.0059	0.0000	0.0000	0.0043	0.0061	0.0000	0.0419
Crotalaria rotundifolia	0.0000	0.0000	0.0000	0.0000	0.0038	0.0035	0.0039	0.0029
Agalinis filifolia	0.0000	0.0053	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Asclepias sp	0.0026	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Carphephorus corymbosus	0.0000	0.0088	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Clitoria mariana	0.0056	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Woodwardia virginica	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Zamia pumila	0.0000	0.0033	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Woody								
Quercus myrtifolia	0.5225	0.6254	0.8850	0.4947	0.8913	0.8120	0.8514	0.8342
Serenoa repens	0.3733	0.3338	0.5421	0.5730	0.4496	0.4946	0.5668	0.4545
Quercus geminata	0.2049	0.1288	0.2116	0.1788	0.3258	0.2940	0.3021	0.3040
Lyonia ferruginea	0.1353	0.2377	0.2861	0.2467	0.1717	0.1981	0.2422	0.2204
Lyonia lucida	0.3038	0.4868	0.3067	0.2024	0.2095	0.1958	0.2120	0.1961
Quercus chapmanii	0.1680	0.1805	0.2191	0.1466	0.2304	0.2098	0.2008	0.1779
Vaccinium myrsinites	0.2186	0.1309	0.0712	0.0000	0.0524	0.0760	0.0794	0.0843
Ilex glabra	0.1136	0.0202	0.0185	0.0188	0.0746	0.1169	0.0873	0.0747
Licania michauxii	0.0267	0.0176	0.0000	0.0000	0.0697	0.0663	0.0246	0.0580
Myrica cerifera	0.0581	0.0500	0.1050	0.0174	0.0382	0.0547	0.0660	0.0512
Quercus minina	0.1165	0.1275	0.0942	0.0000	0.0300	0.0346	0.0617	0.0401
Pinus clausa	0.0000	0.0000	0.0079	0.0096	0.0031	0.0059	0.0170	0.0390
Lyonia fruticosa	0.0839	0.0000	0.0230	0.0510	0.0206	0.0358	0.0380	0.0368
Rhus copallinum	0.0000	0.0000	0.0000	0.0000	0.0394	0.0394	0.0000	0.0301
Gaylussacia nana	0.0115	0.1897	0.0082	0.0000	0.0400	0.0308	0.0188	0.0273
Gaylussacia dumosa	0.1566	0.0781	0.0000	0.0000	0.0360	0.0228	0.0233	0.0250
Smilax auriculata	0.0157	0.0106	0.0000	0.0000	0.0033	0.0208	0.0275	0.0195
Brevaria racemosa	0.0473	0.0128	0.0000	0.0210	0.0239	0.0270	0.0250	0.0178
Pinus elliottii	0.0056	0.0000	0.0121	0.0000	0.0000	0.0029	0.0067	0.0109
Pinus palustris	0.0083	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0060
Quercus nigra	0.0000	0.0000	0.0061	0.0000	0.0000	0.0000	0.0000	0.0000
Quercus sp	0.0086	0.0305	0.0142	0.0000	0.0032	0.0000	0.0000	0.0000
Ceratolia ericoides	0.0402	0.0900	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Osmanthus americanus	0.0000	0.0025	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Salix caroliniana	0.0000	0.0000	0.0000	0.0000	0.0033	0.0030	0.0000	0.0000
Smilax spp	0.0032	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Cladonia evansii	0.0962	0.0638	0.1239	0.0000	0.0000	0.0000	0.0000	0.0000
Opuntia humifusa	0.0028	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000

Table 3-9. Pre- and postburn absolute densities of herb and woody species in 2M in Cedar Key Scrub State Reserve. Absolute densities for control sites 5A and 5D are also shown. d = days. M = months.

	Control		Time after burning					
Species	5A	5D	Pre	12 d	3 M	6 M	9 M	12 M
Herbaceous								
Galactia elliottii	2.46	1.09	0.00	2.48	3.37	0.59	0.40	2.50
Poaceae	0.37	0.14	0.17	0.02	0.55	0.51	0.50	0.77
Solidago odora	0.00	0.00	0.00	0.02	0.14	0.12	0.02	0.35
Woodwardia virginica	0.00	0.00	0.00	0.00	0.30	0.04	0.00	0.21
Galactia mollis	0.00	0.18	0.00	0.12	0.13	0.00	0.00	0.07
Agalinis filifolia	0.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00
Asclepias sp	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Carphephorus corymbosus	0.00	0.19	0.00	0.00	0.00	0.00	0.00	0.00
Clitoria mariana	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Crotalaria rotundifolia	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Zamia pumila	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00
Woody								
Quercus myrtifolia	10.91	12.17	12.13	9.31	29.55	31.68	31.27	21.84
Lyonia ferruginea	2.09	5.06	5.40	2.16	8.84	13.66	14.11	13.12
Gaylussacia nana	0.31	5.30	2.78	5.23	16.54	15.82	10.19	11.54
Quercus geminata	2.90	2.24	6.62	1.82	13.52	13.71	13.18	11.50
Lyonia lucida	7.03	10.70	5.27	4.10	7.12	8.44	8.10	10.95
Vaccinium myrsinites	5.08	2.04	7.02	0.29	2.69	4.29	8.06	9.67
Quercus chapmanii	3.01	2.01	2.02	4.80	8.42	9.97	9.00	6.90
Lyonia fruticosa	1.88	0.00	0.55	0.77	3.33	4.45	5.04	6.20
Serenoa repens	1.14	1.01	1.21	1.64	2.83	3.20	3.25	3.17
Ilex glabra	2.98	0.55	0.74	0.07	3.28	3.33	3.29	2.38
Licania michauxii	0.76	0.20	0.03	0.14	1.37	1.49	0.34	1.69
Brevaria racemosa	0.57	0.05	0.04	0.02	1.06	1.09	1.43	0.85
Myrica cerifera	1.04	0.47	1.01	0.03	0.77	1.05	1.06	0.76
Smilax auriculata	0.12	0.08	0.02	0.00	0.13	0.16	0.11	0.20
Gaylussacia dumosa	5.07	1.37	0.00	0.18	0.29	0.00	0.00	0.17
Quercus minina	2.50	1.70	1.25	0.00	0.03	0.33	0.88	0.11
Rhus copallinum	0.00	0.00	0.00	0.00	0.04	0.01	0.00	0.05
Pinus clausa	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinus elliottii	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinus palustris	0.05	0.00	0.01	0.01	0.01	0.02	0.00	0.00
Quercus nigra	0.00	0.00	0.15	0.02	0.00	0.00	0.00	0.00
Quercus sp	0.07	0.17	0.95	0.03	0.00	0.00	0.00	0.00
Ceratolia ericoides	0.03	0.02	0.03	0.00	0.00	0.00	0.00	0.00
Osmanthus americanus	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00
Salix caroliniana	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Smilax spp	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cladonia evansii	3.51	1.80	7.44	0.00	0.00	0.00	0.00	0.00
Opuntia humifusa	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 3-10. Pre- and postburn absolute frequencies of herb and woody species in 2M in Cedar Key Scrub State Reserve. Absolute frequencies for control sites 5A and 5D are also presented. d = days. M = Months.

	Control			Time after burning				
Species	5A	5D	Pre	12 d	3 M	6 M	9 M	12 M
Herbaceous								
Galactia elliottii	31	29	0	30	40	28	3	36
Poaceae	12	6	9	8	15	14	14	17
Solidago odora	0	0	0	2	6	6	3	8
Galactia mollis	0	1	0	1	2	0	1	3
Woodwardia virginica	0	0	0	0	2	1	0	2
Agalinis filifolia	0	1	0	0	0	0	0	0
Asclepias sp	1	0	0	0	0	0	0	0
Carphephorus corymbosus	0	1	0	0	0	0	0	0
Clitoria mariana	2	0	0	0	0	0	0	0
Crotalaria rotundifolia	0	0	0	0	0	0	0	0
Zamia pumila	0	1	0	0	0	0	0	0
Woody								
Quercus myrtifolia	34	44	48	44	44	45	46	44
Quercus geminata	29	34	43	23	44	43	43	44
Gaylussacia nana	1	29	9	30	38	38	33	39
Vaccinium myrsinites	34	27	38	6	32	37	40	39
Serenoa repens	27	29	35	36	36	36	36	37
Quercus chapmanii	32	35	34	28	33	37	36	34
Myrica cerifera	14	12	19	1	15	18	17	19
Lyonia ferruginea	19	34	20	17	18	19	18	18
Lyonia lucida	28	38	17	12	16	16	16	17
Quercus minina	26	40	23	0	3	15	23	12
Licania michauxii	5	6	1	3	11	11	5	10
Lyonia fruticosa	13	0	4	3	5	6	6	6
Brevaria racemosa	8	2	2	2	5	5	5	5
Ilex glabra	13	4	4	2	5	5	5	5
Smilax auriculata	5	4	1	0	4	4	4	4
Gaylussacia dumosa	24	22	0	2	4	2	0	3
Rhus copallinum	0	0	0	0	3	1	0	3
Pinus palustris	3	0	1	1	1	1	0	1
Pinus clausa	0	0	0	0	0	0	0	0
Pinus elliottii	2	0	0	0	0	0	0	0
Quercus nigra	0	0	6	1	0	0	0	0
Quercus sp	3	12	21	1	0	0	0	0
Ceratolia ericoides	3	6	4	0	0	1	1	0
Osmanthus americanus	0	1	0	0	0	0	0	0
alix caroliniana	0	0	0	0	0	0	0	0
Smilax spp	1	0	0	0	0	0	0	0
Cladonia evansii	8	12	21	0	0	0	0	0
Opuntia humifusa	1	0	0	0	0	0	0	0

Table 3-11. Pre- and postburn absolute mean percent cover of herb and woody species in 2M in Cedar Key Scrub State Reserve. Absolute mean percent cover for control sites 5A and 5D are also shown. d = days. M = Months.

	Control		Time after burning					
Species	5A	5D	Pre	12 d	3 M	6 M	9 M	12 M
Herbaceous								
Galactia elliottii	2.06	1.76	0.00	0.04	1.61	0.08	0.13	1.23
Poaceae	0.47	0.00	0.15	0.00	0.16	0.39	0.33	0.55
Solidago odora	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.30
Galactia mollis	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.11
Agalinis filifolia	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Asclepias sp	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Carphephorus corymbosus	0.00	0.26	0.00	0.00	0.00	0.00	0.00	0.00
Clitoria mariana	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Crotalaria rotundifolia	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Woodwardia virginica	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.00
Zamia pumila	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.00
Woody								
Serenoa repens	23.95	24.39	24.40	1.00	13.34	20.07	21.57	21.64
Quercus myrtifolia	20.07	27.36	22.21	0.73	14.17	12.47	14.48	18.68
Lyonia ferruginea	4.22	5.70	6.27	0.00	3.24	5.98	5.26	11.18
Quercus geminata	6.75	0.64	4.96	0.15	3.98	5.13	4.63	8.58
Quercus chapmanii	2.89	5.97	2.61	0.62	2.09	2.35	2.97	4.07
Vaccinium myrsinites	3.55	2.80	2.80	0.00	1.06	1.20	1.43	2.76
Lyonia lucida	8.94	18.00	7.30	0.00	1.81	3.18	3.11	2.62
Gaylussacia nana	0.28	1.60	0.63	0.15	1.70	1.84	0.61	1.75
Lyonia fruticosa	1.48	0.00	0.02	0.00	0.10	0.30	0.66	0.77
Ilex glabra	2.28	0.00	0.44	0.00	0.90	0.54	0.46	0.74
Licania michauxii	0.04	0.00	0.00	0.00	0.20	0.07	0.00	0.36
Myrica cerifera	0.39	1.32	1.36	0.00	0.06	0.45	0.19	0.32
Pinus clausa	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinus elliottii	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinus palustris	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Quercus minina	0.59	0.27	0.00	0.00	0.00	0.00	0.03	0.00
Quercus nigra	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00
Quercus sp	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Brevaria racemosa	1.44	0.72	0.20	0.00	0.00	0.09	0.14	0.00
Ceratolia ericoides	2.71	7.49	3.53	0.00	0.00	0.22	0.14	0.00
Gaylussacia dumosa	0.43	0.06	0.00	0.00	0.02	0.00	0.00	0.00
Osmanthus americanus	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rhus copallinum	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Salix caroliniana	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Smilax auriculata	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Smilax spp	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cladonia evansii	1.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Opuntia humifusa	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 3-12. Pre- and postburn absolute importance values of herb and woody species in 2M in Cedar Key Scrub State Reserve. Absolute importance values for control sites 5A and 5D are also displayed. d = days. M = months.

	Control		Time after burning					
Species	5A	5D	Pre	12 d	3 M	6 M	9 M	12 M
Herbaceous								
Galactia elliottii	0.1457	0.1055	0.0000	0.1984	0.1693	0.0780	0.0141	0.1276
Poaceae	0.0418	0.0164	0.0280	0.0302	0.0466	0.0475	0.0486	0.0559
Solidago odora	0.0000	0.0000	0.0000	0.0080	0.0165	0.0172	0.0084	0.0268
Galactia mollis	0.0000	0.0059	0.0000	0.0072	0.0076	0.0000	0.0027	0.0094
Woodwardia virginica	0.0000	0.0000	0.0000	0.0000	0.0090	0.0029	0.0000	0.0069
Agalinis filifolia	0.0000	0.0053	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Asclepias sp	0.0026	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Carphephorus corymbosus	0.0000	0.0088	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Clitoria mariana	0.0056	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Crotalaria rotundifolia	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Zamia pumila	0.0000	0.0033	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Woody								
Quercus myrtifolia	0.5225	0.6254	0.6288	0.7094	0.7121	0.6230	0.6674	0.5620
Serenoa repens	0.3733	0.3338	0.4283	0.5528	0.4175	0.4904	0.5130	0.4062
Quercus geminata	0.2049	0.1288	0.2928	0.1955	0.3298	0.3247	0.3195	0.3300
Lyonia ferruginea	0.1353	0.2377	0.2293	0.1267	0.2028	0.2787	0.2708	0.3166
Gaylussacia nana	0.0115	0.1897	0.0809	0.3220	0.2923	0.2696	0.1933	0.2280
Vaccinium myrsinites	0.2186	0.1309	0.2592	0.0308	0.1303	0.1541	0.2078	0.2234
Quercus chapmanii	0.1680	0.1805	0.1568	0.4753	0.2108	0.2253	0.2328	0.2023
Lyonia lucida	0.3038	0.4868	0.2327	0.1654	0.1492	0.1735	0.1726	0.1803
Lyonia fruticosa	0.0839	0.0000	0.0203	0.0338	0.0468	0.0599	0.0737	0.0838
Myrica cerifera	0.0581	0.0500	0.0843	0.0046	0.0466	0.0634	0.0594	0.0577
Licania michauxii	0.0267	0.0176	0.0031	0.0152	0.0453	0.0424	0.0167	0.0451
Ilex glabra	0.1136	0.0202	0.0292	0.0095	0.0642	0.0519	0.0516	0.0446
Quercus minina	0.1165	0.1275	0.0811	0.0000	0.0079	0.0412	0.0714	0.0302
Brevaria racemosa	0.0473	0.0128	0.0084	0.0080	0.0228	0.0239	0.0291	0.0203
Smilax auriculata	0.0157	0.0106	0.0029	0.0000	0.0113	0.0116	0.0119	0.0116
Gaylussacia dumosa	0.1566	0.0781	0.0000	0.0127	0.0133	0.0051	0.0000	0.0089
Rhus copallinum	0.0000	0.0000	0.0000	0.0000	0.0080	0.0026	0.0000	0.0078
Pinus palustris	0.0083	0.0000	0.0027	0.0040	0.0026	0.0027	0.0000	0.0024
Pinus clausa	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Pinus elliottii	0.0056	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Quercus nigra	0.0000	0.0000	0.0183	0.0043	0.0000	0.0000	0.0000	0.0000
Quercus sp	0.0086	0.0305	0.0706	0.0046	0.0000	0.0000	0.0000	0.0000
Ceratolia ericoides	0.0402	0.0900	0.0566	0.0000	0.0000	0.0000	0.0000	0.0000
Osmanthus americanus	0.0000	0.0025	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Salix caroliniana	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Smilax spp	0.0032	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Cladonia evansii	0.0962	0.0638	0.1869	0.0000	0.0000	0.0000	0.0000	0.0000
Opuntia humifusa	0.0028	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000

Table 3-13. Density of ramets in 5C after prescribed burning in Cedar Key Scrub State Reserve.

	Time after burning				
	11	3	6	9	12
Species	days	Months	Months	Months	Months
Herbaceous					
Poaceae	0.09	0.61	1.15	0.9800	1.9000
Solidago odora	0.00	0.07	0.05	0.0500	1.8000
Galactia elliottii	0.00	2.66	1.26	0.0000	1.3500
Gallactia moilis	0.00	0.04	0.05	0.0000	1.2200
Crotalaria rotundifolia	0.00	0.08	0.07	0.0700	0.0300
Agalinis filifolia	0.00	0.00	0.00	0.0000	0.0000
Asclepias sp	0.00	0.00	0.00	0.0000	0.0000
Carphephorus corymbosus	0.00	0.00	0.00	0.0000	0.0000
Clitoria mariana	0.00	0.00	0.00	0.0000	0.0000
Woodwardia virginica	0.00	0.00	0.00	0.0000	0.0000
Zamia pumila	0.00	0.00	0.00	0.0000	0.0000
Woody					
Quercus myrtifolia	0.00	44.13	41.98	46.2500	40.5300
Quercus geminata	0.00	12.82	12.28	12.8600	11.1900
Lyonia ferruginea	0.00	7.74	9.81	10.6400	11.0100
Lyonia lucida	0.00	7.68	8.28	8.3200	8.3300
Quercus chapmanii	0.00	9.70	8.28	7.0400	6.1100
Serenoa repens	1.80	3.31	4.32	4.3400	4.9400
Ilex glabra	0.00	4.57	4.84	4.6700	3.4700
Licania michauxii	0.00	2.65	2.43	0.9000	2.3400
Vaccinum myrsinites	0.00	0.97	1.43	1.4400	1.6900
Lyonia fruticosa	0.00	1.09	1.70	2.0600	1.5500
Gaylussacia dumosa	0.00	1.69	1.41	1.4600	1.4000
Gaylussacia nana	0.00	1.32	1.19	0.2800	1.1300
Brevaria racemosa	0.00	1.89	1.97	1.6200	1.0200
Myrica cerifera	0.00	0.42	0.76	0.6800	0.4900
Rhus copallinum	0.00	0.25	0.28	0.0000	0.3100
Pinus clausa	0.00	0.00	0.03	0.0800	0.2500
Smilax auriculata	0.00	0.03	0.11	0.1700	0.1200
Quercus minina	0.00	0.30	0.38	0.3500	0.0900
Pinus palustris	0.00	0.00	0.00	0.0000	0.0800
Pinus elliottii	0.00	0.00	0.01	0.0200	0.0500
Quercus nigra	0.00	0.00	0.00	0.0000	0.0000
Quercus sp	0.00	0.02	0.00	0.0000	0.0000
Ceratolia ericoides	0.00	0.00	0.00	0.0000	0.0000
Osmanthus americanus	0.00	0.00	0.00	0.0000	0.0000
Salix caroliniana	0.00	0.03	0.02	0.0000	0.0000
Smilax spp	0.00	0.00	0.00	0.0000	0.0000

Table 3-14. Density of ramets in 2M after prescribed burning in Cedar Key Scrub State Reserve.

	Time after burning				
Species	12 days	3 Months	6 Months	9 Months	12 Months
Herbaceous					
Galactia elliottii	2.48	3.37	0.59	0.4000	2.5000
Poaceae	0.02	0.55	0.51	0.5000	0.7700
Solidago odora	0.02	0.08	0.07	0.0200	0.3500
Woodwardia virginica	0.00	0.30	0.04	0.0000	0.2100
Gallactia moilis	0.12	0.13	0.00	0.0000	0.0700
Agalinis filifolia	0.00	0.00	0.00	0.0000	0.0000
Asclepias sp	0.00	0.00	0.00	0.0000	0.0000
Carphephorus corymbosus	0.00	0.00	0.00	0.0000	0.0000
Clitoria mariana	0.00	0.00	0.00	0.0000	0.0000
Crotalaria rotundifolia	0.00	0.00	0.00	0.0000	0.0000
Zamia pumila	0.00	0.00	0.00	0.0000	0.0000
Woody					
Quercus myrtifolia	7.85	28.08	30.18	29.7700	20.1700
Lyonia ferruginea	1.00	7.68	12.50	12.9500	11.7900
Gaylussacia nana	5.23	16.54	15.82	10.1900	11.5400
Quercus geminata	1.50	13.20	13.39	12.8600	11.0600
Lyonia lucida	3.42	6.47	7.79	7.4500	10.3000
Vaccinum myrsinites	0.29	2.69	4.29	8.0600	9.6700
Quercus chapmanii	4.52	8.14	9.67	8.7000	6.5500
Lyonia fruticosa	0.74	3.30	4.42	5.0100	6.1700
Serenoa repens	1.64	2.83	3.20	3.2500	3.1700
Ilex glabra	0.00	3.21	3.26	3.2200	2.3100
Licania michauxii	0.14	1.37	1.49	0.3400	1.6900
Brevaria racemosa	0.00	1.04	1.07	1.4100	0.8200
Myrica cerifera	0.03	0.77	1.05	1.0600	0.7600
Smilax auriculata	0.00	0.13	0.16	0.1100	0.2000
Gaylussacia dumosa	0.18	0.29	0.00	0.0000	0.1700
Quercus minina	0.00	0.03	0.33	0.8800	0.1100
Rhus copallinum	0.00	0.04	0.01	0.0000	0.0500
Pinus clausa	0.00	0.00	0.00	0.0000	0.0000
Pinus elliottii	0.00	0.00	0.00	0.0000	0.0000
Pinus palustris	0.00	0.00	0.02	0.0000	0.0000
Quercus nigra	0.02	0.00	0.00	0.0000	0.0000
Quercus sp	0.03	0.00	0.00	0.0000	0.0000
Ceratolia ericoides	0.00	0.00	0.00	0.0000	0.0000
Osmanthus americanus	0.00	0.00	0.00	0.0000	0.0000
Salix caroliniana	0.00	0.00	0.00	0.0000	0.0000
Smilax spp	0.00	0.00	0.00	0.0000	0.0000

Table 3-15. Tree mortality in quadrats and in grids 5C and 2M in Cedar Key Scrub State Reserve.

		Trees Before Burning		Dead Trees After Burning		Mortality Percentage	
		5C	2M	5C	2M	5C	2M
Quadrats	*Pinus clausa*	1	0	0	0	0.0	0.0
	Pinus elliottii	3	0	3	0	100.0	0.0
	Pinus palustris	0	1	0	1	0.0	100.0
	Quercus myrtifolia	43	21	0	3	0.0	14.3
	Quercus chapmanii	13	4	3	0	23.1	0.0
	Quercus geminata	15	9	0	0	0.0	0.0
	Lyonia ferruginea	35	53	4	0	11.4	0.0
	Ceratolia ericoides	0	1	0	1	0.0	100.0
	Subtotal	110	89	10	5	9.1	5.6
	Total	199		15		7.5	
Grids	*Pinus clausa*	18	0	14	0	77.8	0.0
	Pinus elliottii	108	1	96	1	88.9	0.0
	Pinus palustris	47	53	44	53	93.6	100.0
	Subtotal	173	54	154	54	89.0	100.0
	Total	227		208		91.6	

Table 3-16. Coefficient of determination (r^2) resulting from Detrended Correspondence Analysis of the pre- and postburn sites of a long-unburned scrubby flatwoods in Cedar Key Scrub State Reserve.

Site	Data	Axis	r squared Increment	r squared Cumulative
5C Woody	Absolute Density	1	0.8440	0.8440
		2	0.0760	0.9200
		3	0.0300	0.9500
	Absolute Mean % Cover	1	0.6260	0.6260
		2	0.3600	0.9860
		3	-0.0520	0.9340
	Relativized Density	1	0.7260	0.7260
		2	0.2320	0.9590
		3	0.0270	0.9860
	Relativized Mean % Cover	1	0.8790	0.8790
		2	0.0980	0.9770
		3	-0.0040	0.9720
2M Woody	Absolute Density	1	0.9290	0.9290
		2	0.0430	0.9720
		3	0.0050	0.9770
	Absolute Mean % Cover	1	0.2570	0.2570
		2	0.6590	0.9160
		3	0.0380	0.9540
	Relativized Density	1	0.9090	0.9090
		2	0.0560	0.9650
		3	0.0230	0.9880
	Relativized Mean % Cover	1	0.5480	0.5480
		2	0.3830	0.9310
		3	-0.0090	0.9220

Table 3-17. Summary statistics of the Multi-response Permutation Procedure for woody absolute densities and mean percent cover between control and treatment sites at preburn and 12 months postburn in Cedar Key Scrub State Reserve. Results are given for Euclidean and Sorensen distances.

Variable	Distance	Observed d	Expected	Variance	Skewness	T	p
Absolute Density	Euclidean	46.9029	50.8919	0.0159	-0.7659	-31.6566	< 0.001
	Sorensen	0.7039	0.7569	0.0000	-0.4809	-35.9750	< 0.001
Absolute Cover	Euclidean	73.2009	73.6672	0.0611	-0.8182	-1.8872	0.0469
	Sorensen	0.7608	0.7703	0.0000	-0.7717	-4.0362	0.0014

Code: Observed = Observed delta. Expected = Expected delta. Delta is the weighted average distance within-group distance. T is the value of the T test statistics.

Table 3-18. Multiple comparison for absolute densities and mean percent cover between control and treatment sites at 12 months postburn and between preburn and 12 months postburn values of treatment sites in Cedar Key Scrub State Reserve.

		Absolute Density		Absolute Cover	
	Sites	T	p-value	T	p-value
Euclidean	5A vs 5C post	-17.5538	<0.0001	-0.358058	0.26219348
	5A vs 2M post	-12.9969	<0.0001	-0.449258	0.24910358
	5D vs 5C post	-22.5284	<0.0001	-3.657711	0.00845920
	5D vs 2M post	-24.4070	<0.0001	-4.644292	0.00223107
	5C pre vs 5C post	-20.0190	<0.0001	0.151180	0.42555303
	2M pre vs 2M post	-12.0040	<0.0001	-0.018410	0.38794972
Sorensen	5A vs 5C post	-17.9852	<0.0001	-1.145694	0.11978385
	5A vs 2M post	-13.8534	<0.0001	-1.447951	0.08799726
	5D vs 5C post	-26.5362	<0.0001	-4.079046	0.00532423
	5D vs 2M post	-29.5114	<0.0001	-6.554787	0.00020113
	5C pre vs 5C post	-15.4409	<0.0001	0.224953	0.45691273
	2M pre vs 2M post	-10.0212	<0.0001	-0.622968	0.21616540

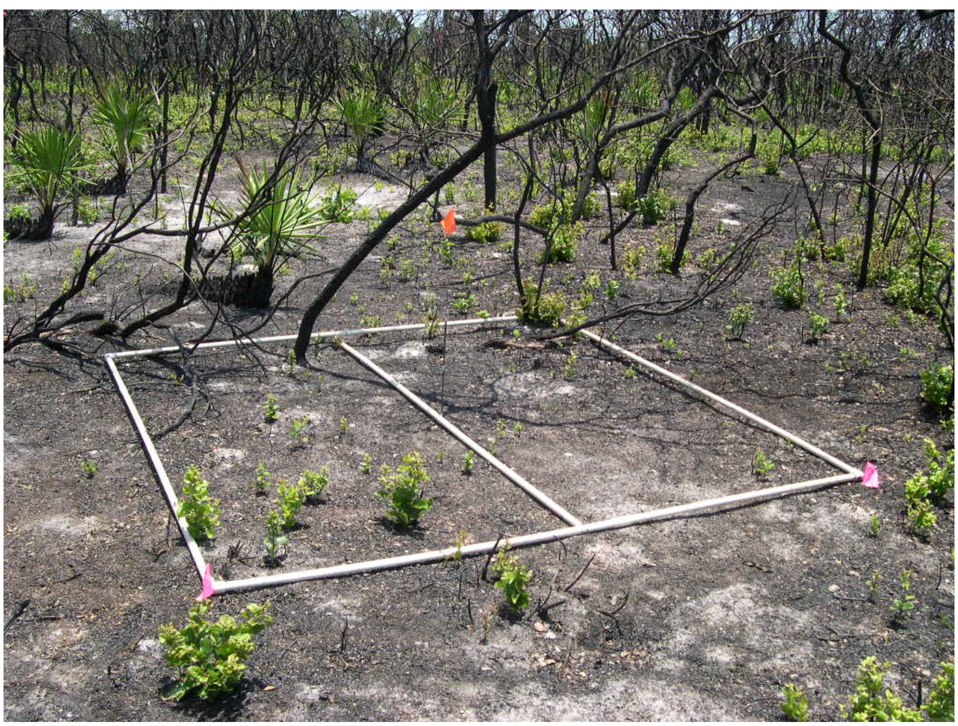
T is the value of the T test statistics.

Table 3-19. Comparison among several studies and Cedar Key Scrub State Reserve (CKSRR) regarding common variables measured in each research. The data presented for bareground and for plant species is mean percent cover. Data for Schmalzer and Hinkle's study average all strata. Data for CKSSR is the average of the two treatment sites.

	Schmalzer & Hinkle 1992 Oak - saw palmetto scrub				Abrahamson 1996 Scrubby flatwoods			Greenberg 2003 Sand pine scrub				Silva-Lugo Scrubby flatwoods			
	Pre	6 M	12 M	Rec	Pre	12 M	Rec	Pre	5 M	16 M	Rec	Pre	6 M	12 M	Rec
Bareground	0.00	22.90	14.60					0.00	15.00	25.00		1.00	3.80	6.80	
Mean height (m)	1.08	0.32	0.50	7.1				1.00	1.00	0.90	0.4	3.53	0.97	1.08	
Species richness	8.50	10.50	10.40		18.60	13.50		10.00	12.30	14.80		18.50	22.00	22.50	
G. elliottii	0.00	1.70	0.00					0.00	0.00	0.00		0.00	0.10	0.87	
Q. myrtifolia	36.00	17.70	18.40	5.0				58.73	22.67	39.59	1.3	25.32	14.65	21.82	1.0
Q. geminata	15.30	11.70	10.60	5.0	4.80	4.80	3.0	2.67	1.14	2.23	1.6	3.83	4.82	8.11	1.0
Q. chapmanii	6.70	4.30	4.20	5.0	15.20	15.00	3.0	0.98	0.33	0.93	1.3	2.68	2.41	3.34	1.0
L. ferruginea								0.84	0.20	1.10	1.0	5.79	3.91	7.14	1.0
L. lucida	17.60	10.20	13.80	4.5	2.75	2.90	3.0					5.82	2.76	3.10	
S. repens	31.80	18.00	30.40	1.3	21.70	18.00	2.0	0.40	0.33	0.30	1.8	26.53	19.70	22.32	
V. myrsinites	1.50	2.30	2.40	0.5	0.52	1.00	3.0					1.59	0.77	1.67	1.0

The citation for Abrahamson 1996 is Abrahamson & Abrahamson 1996. Codes: Pre = preburn. M = months. Rec. = recovery time (years). Empty cells mean data not available.

A



B

Figure 3-1. Quadrats in Cedar Key Scrub State Reserve. A) Quadrat placement in the scrub. B) Location of flags in a quadrat.

A

B

Figure 3-2. Ramets of *Quercus myrtifolia* in Cedar Key Scrub State Reserve. A) Twelve days after burning. B) Twenty nine days postburn.

A

B

Figure 3-3. Ramets of *Quercus chapmanii* in Cedar Key Scrub State Reserve. A) Twelve days after burning. B) Twenty nine days postburn.

A

B

Figure 3-4. Ramets of *Lyonia ferruginea* in Cedar Key Scrub State Reserve. A) Twelve days after burning. B) One year postburn.

A

B

Figure 3-5. Ramets of *Gaylussacia nana* in Cedar Key Scrub State Reserve. A) Apparently, there are three individuals. B) These are sprouts from the same root.

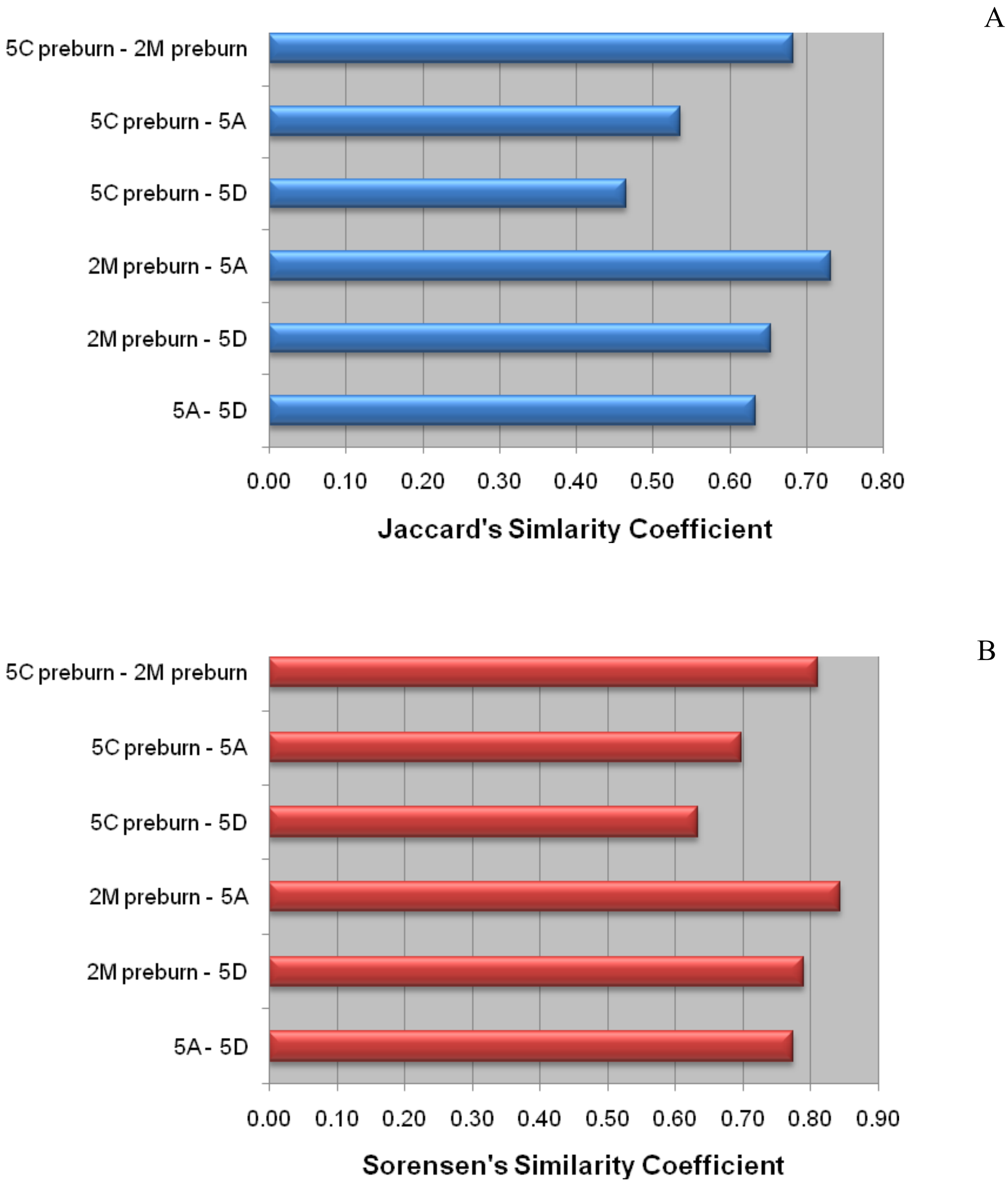


Figure 3-6. Similarity coefficients among the four study sites in Cedar Key Scrub State Reserve. A) Jaccard's similarity coefficient. B) Sorensen's similarity coefficient.

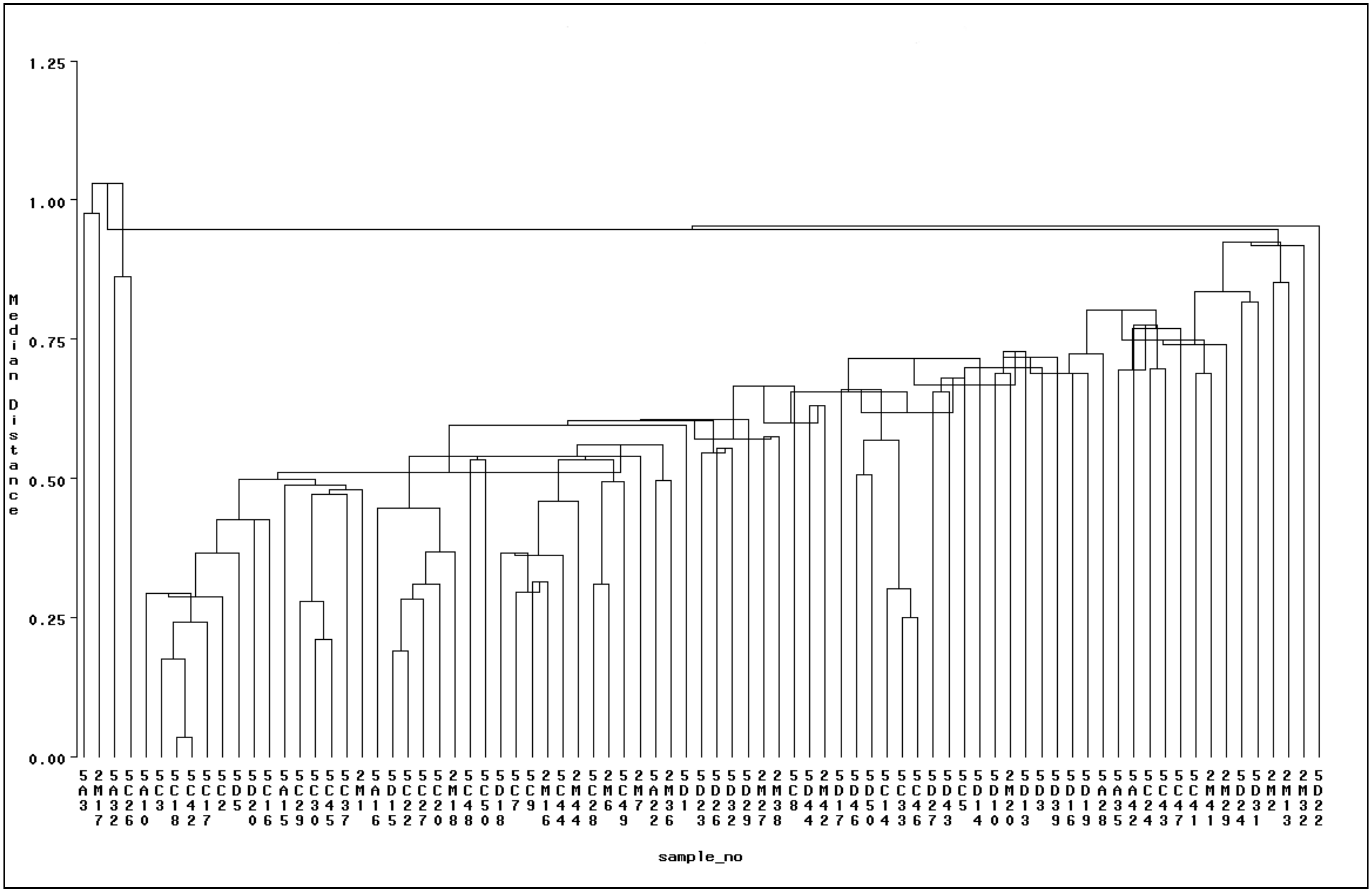


Figure 3-7. Median linkage dendrogram for herb and woody species in treatment and control sites under preburn conditions in Cedar Key Scrub State Reserve.

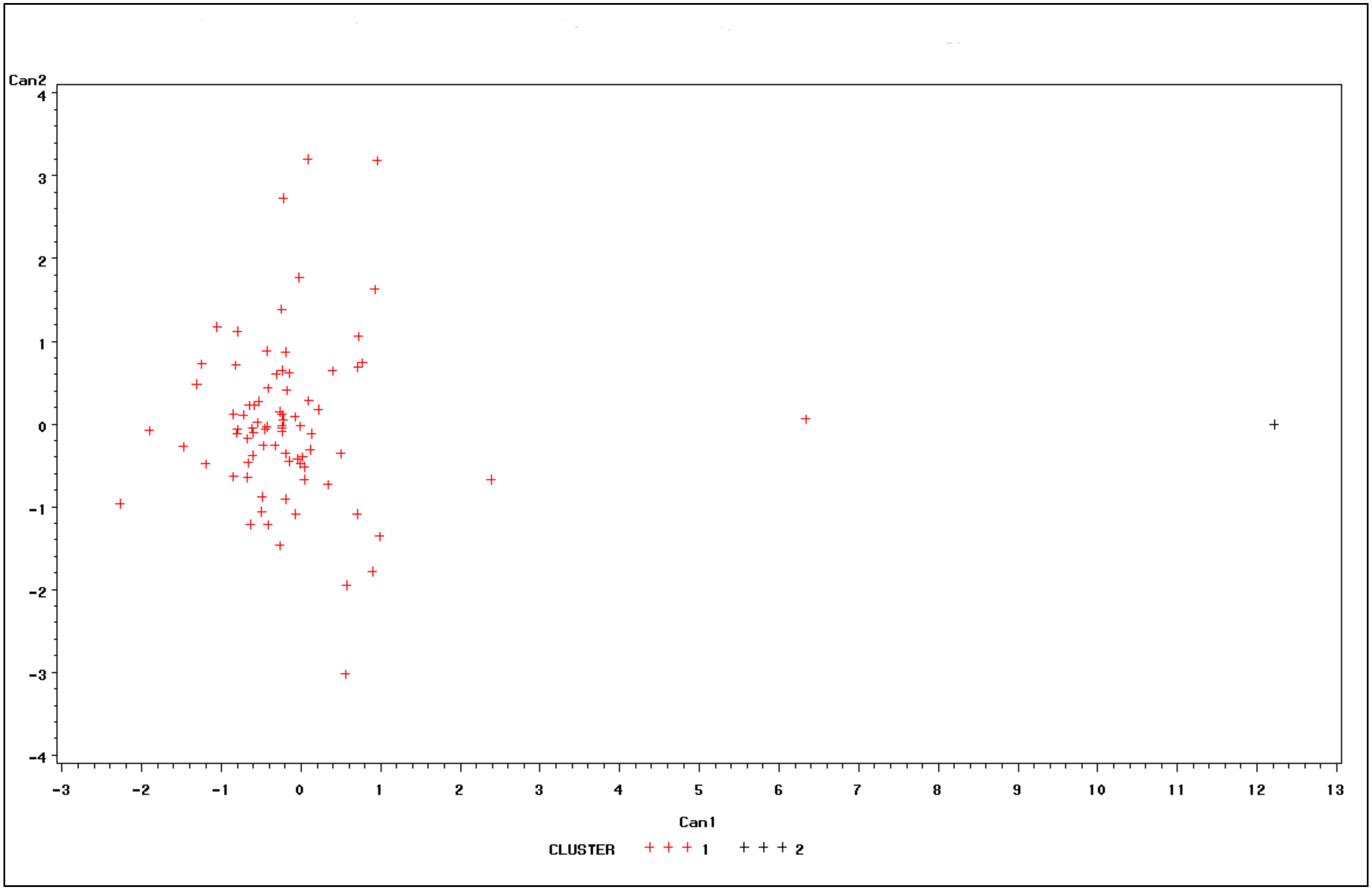


Figure 3-8. Scatterplot of the first two canonical axes corresponding to the cluster analysis with Median linkage fusion method for herb and woody species in treatment and control sites under preburn conditions in Cedar Key Scrub State Reserve.

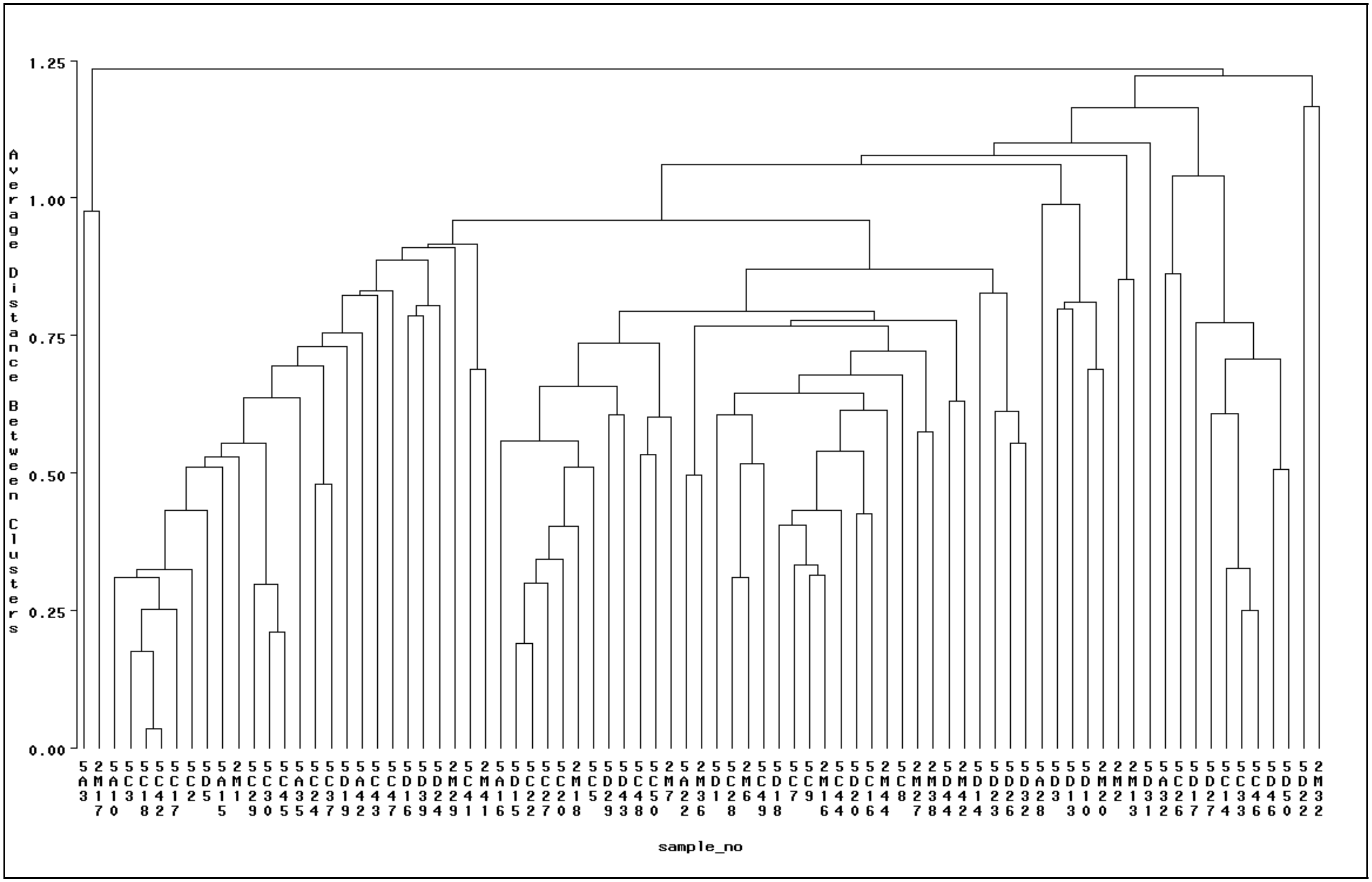


Figure 3-9. Average linkage dendrogram for herb and woody species in treatment and control sites under preburn conditions in Cedar Key Scrub State Reserve.

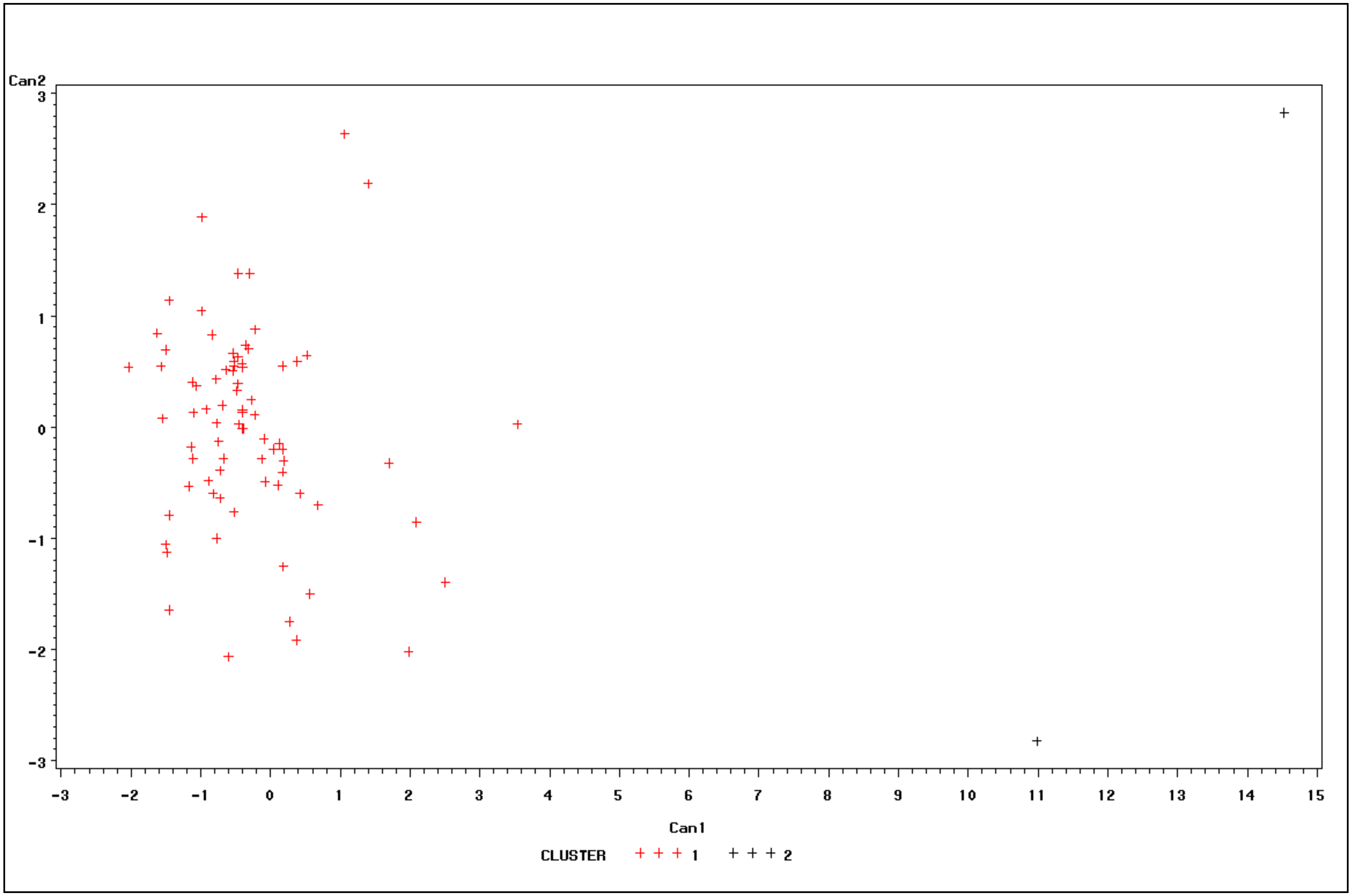


Figure 3-10. Scatterplot of the first two canonical axes corresponding to the cluster analysis with Average linkage fusion method for herb and woody species in treatment and control sites under preburn conditions in Cedar Key Scrub State Reserve.

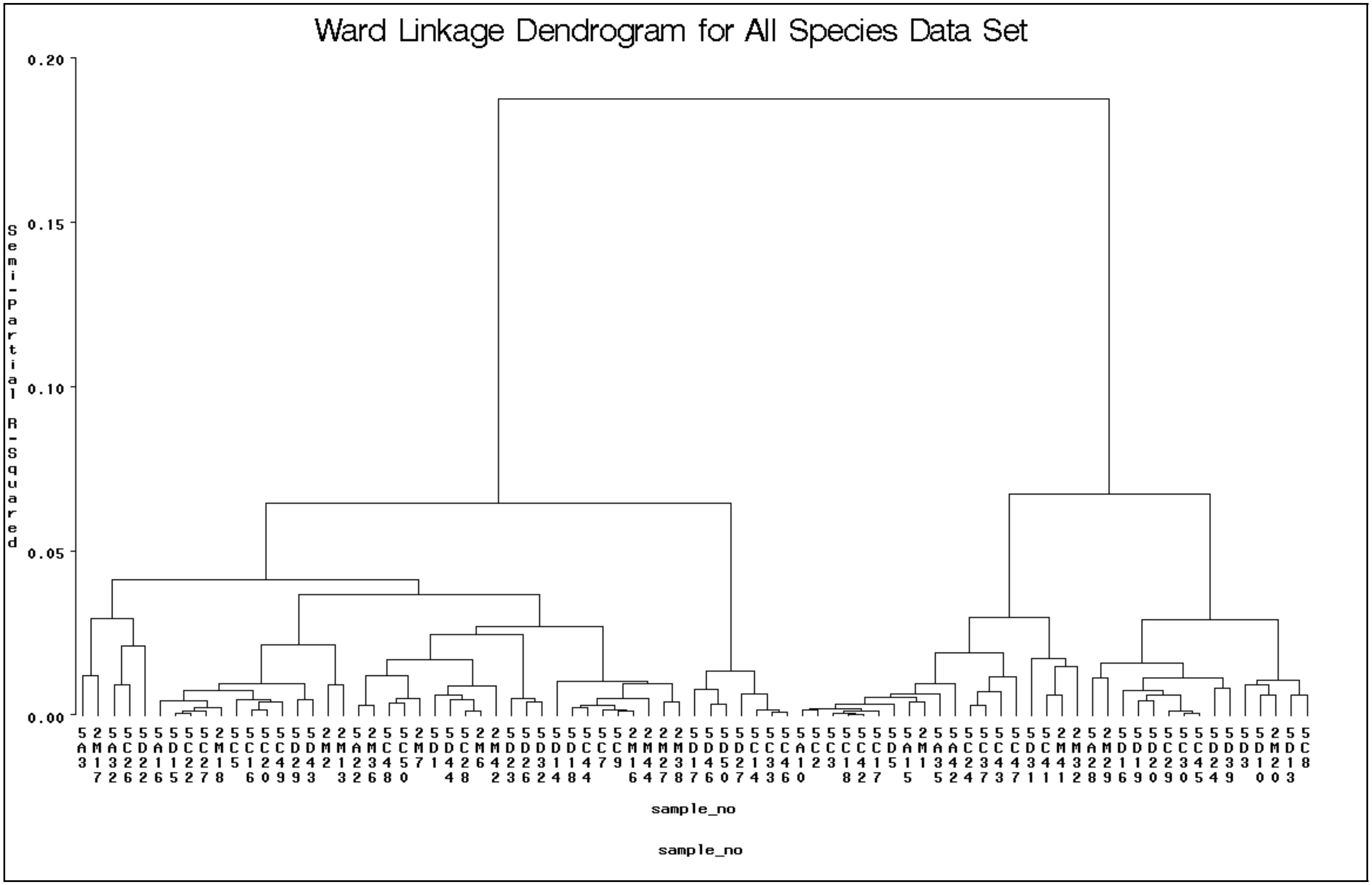


Figure 3-11. Ward’s minimum-variance linkage dendrogram for herb and woody species in treatment and control sites under preburn conditions in Cedar Key Scrub State Reserve.

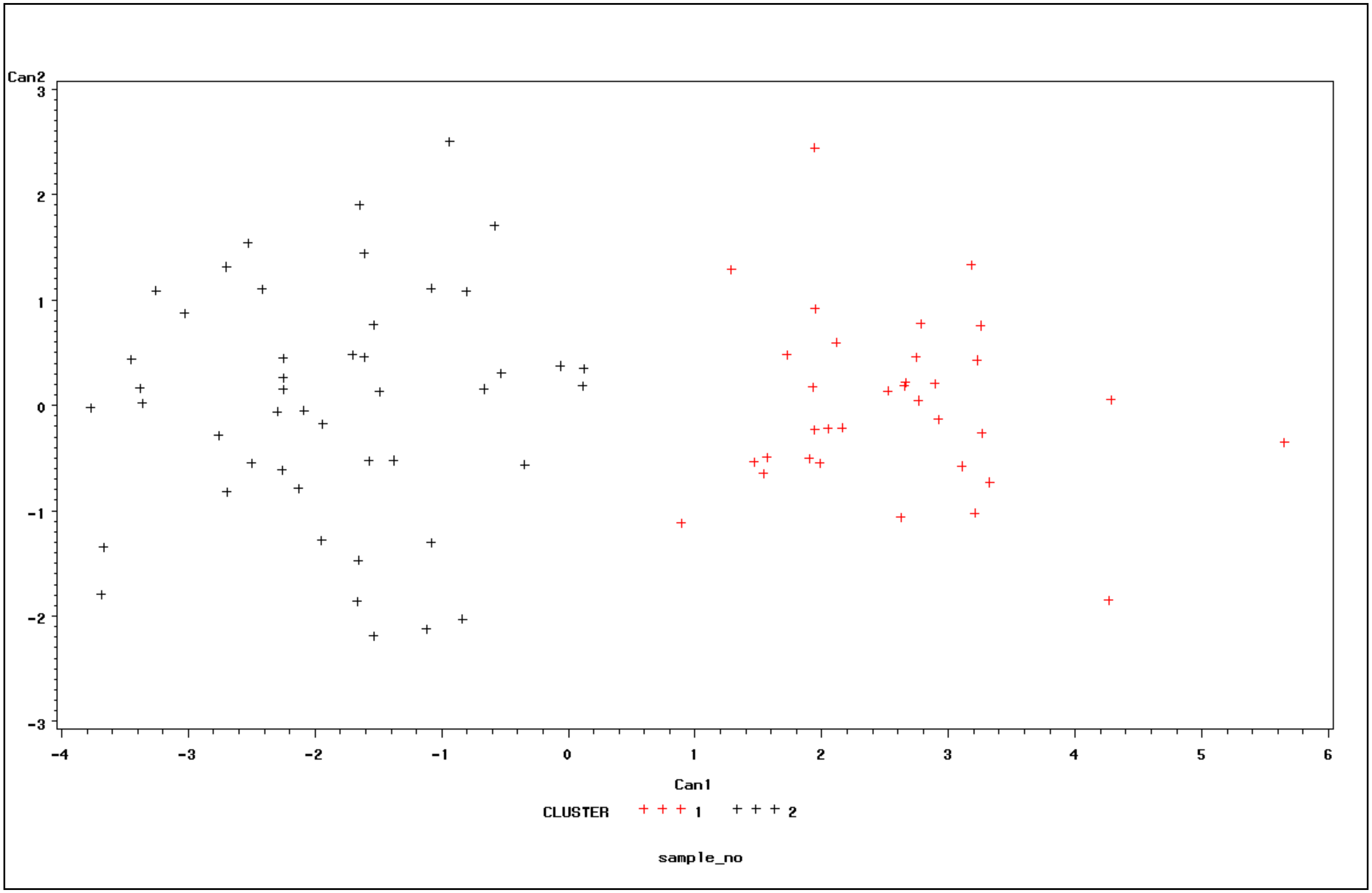


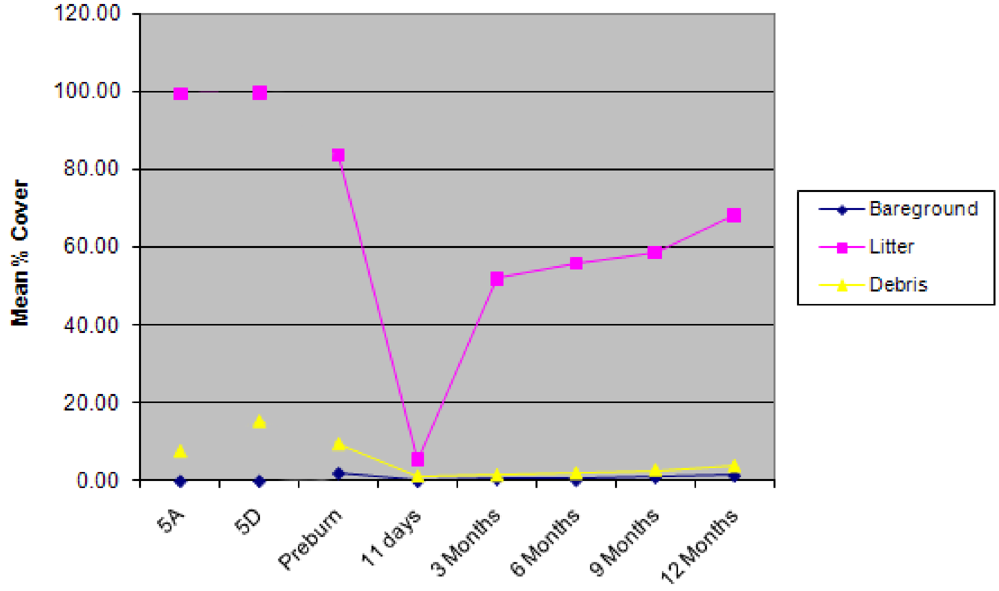
Figure 3-12. Scatterplot of the first two canonical axes corresponding to the cluster analysis with Ward's minimum-variance linkage fusion method for herb and woody species in treatment and control sites under preburn conditions in Cedar Key Scrub State Reserve.

Serenoa repens	Sites	5A	5C pre	2M pre	5D
	Medians	4	4	3	3

Vacinium myrsinites	Sites	2M pre	5A	5D	5C pre
	Medians	10	5	3	2

Figure 3-13. Duncan's multiple comparisons for the median abundances of *Serenoa repens* and *Vaccinium myrsinites* among treatment and control sites in Cedar Key Scrub State Reserve.

A



B

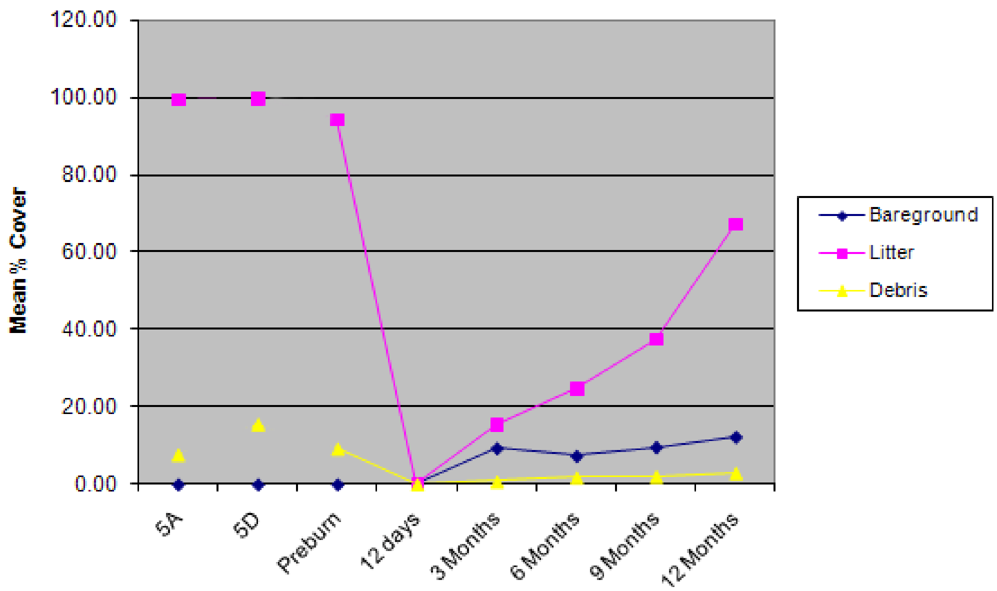


Figure 3-14. Preburn and postburn mean percent cover of bareground, litter, and debris in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M. Control values for 5A and 5D are included.

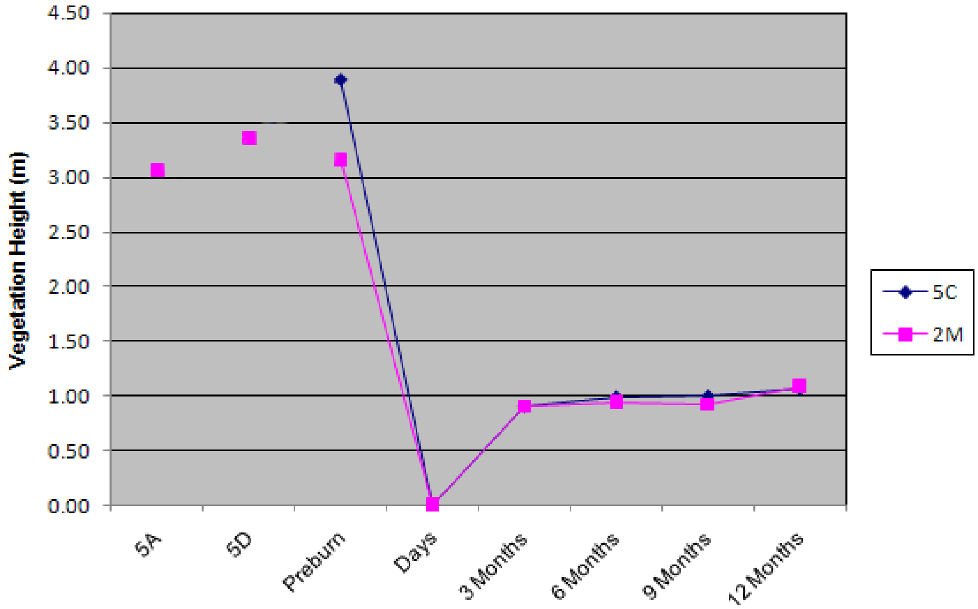
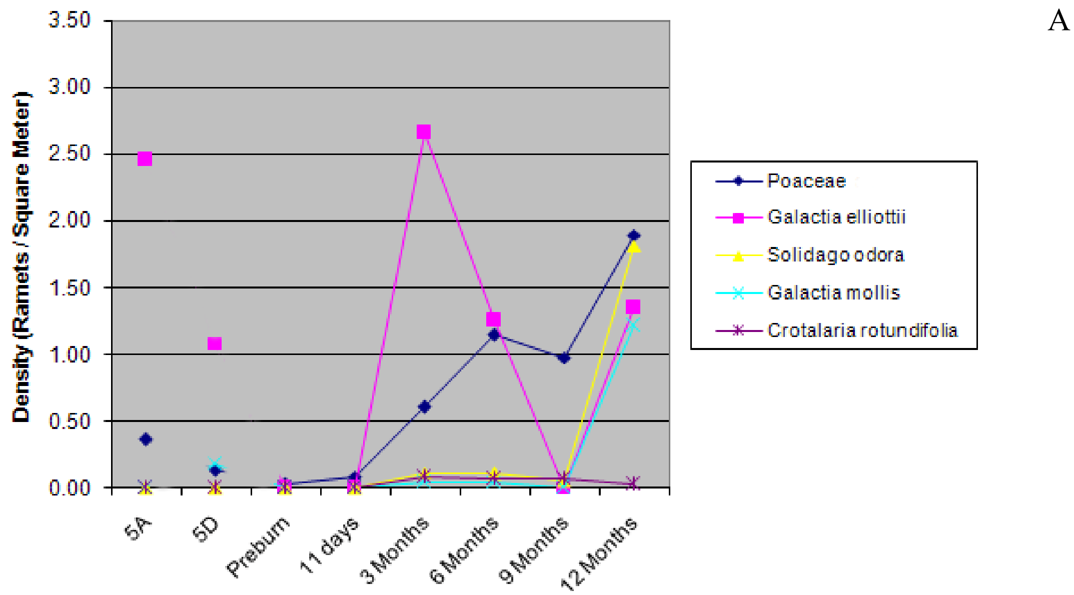


Figure 3-15. Preburn and postburn vegetation height in 5C and 2M in Cedar Key Scrub State Reserve. Control values for 5A and 5D are included.



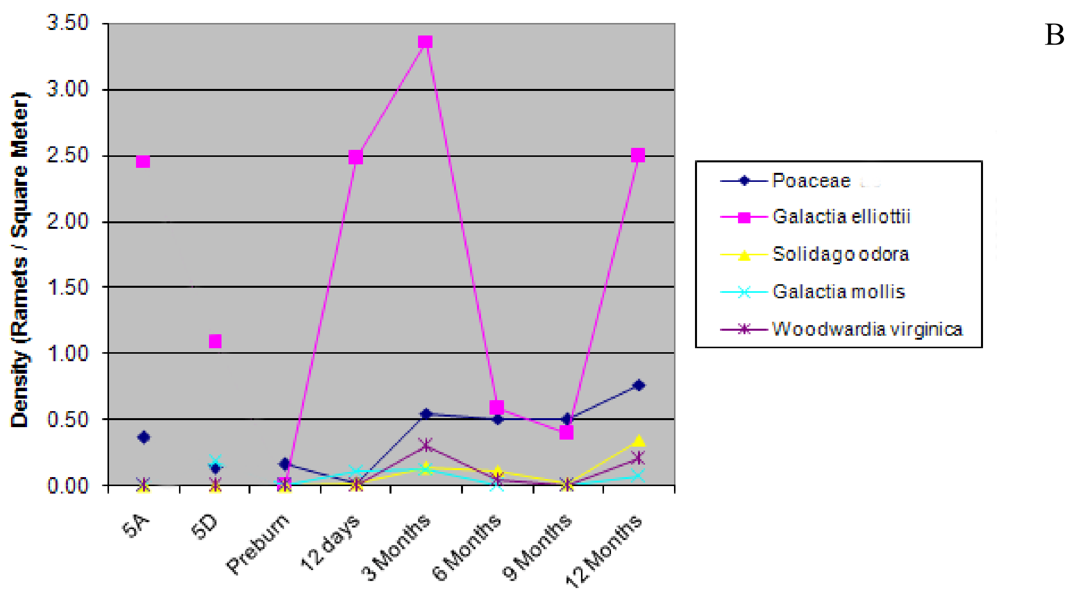
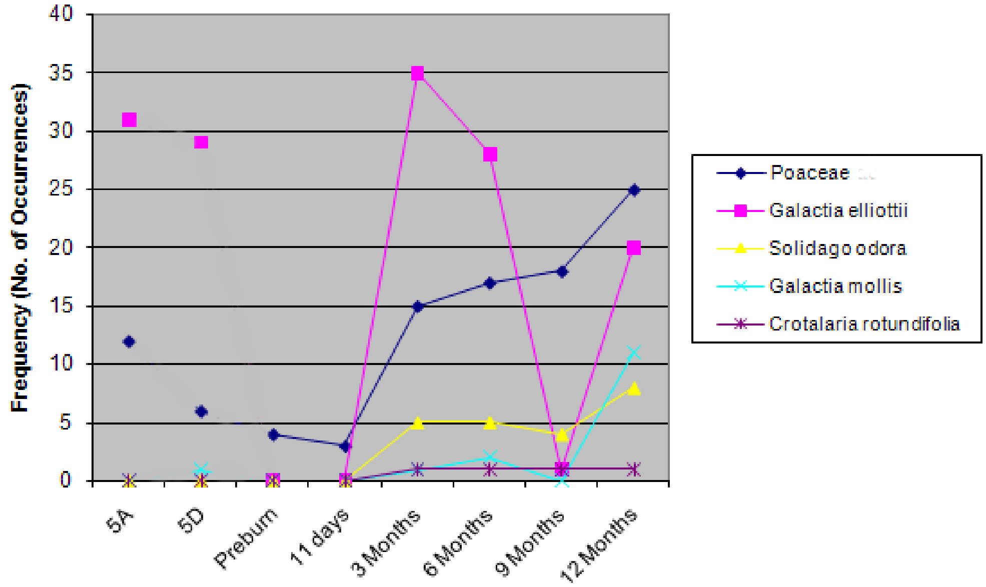


Figure 3-16. Preburn and postburn absolute densities of the most abundance herb species in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M. Control values for 5A and 5D are included

A



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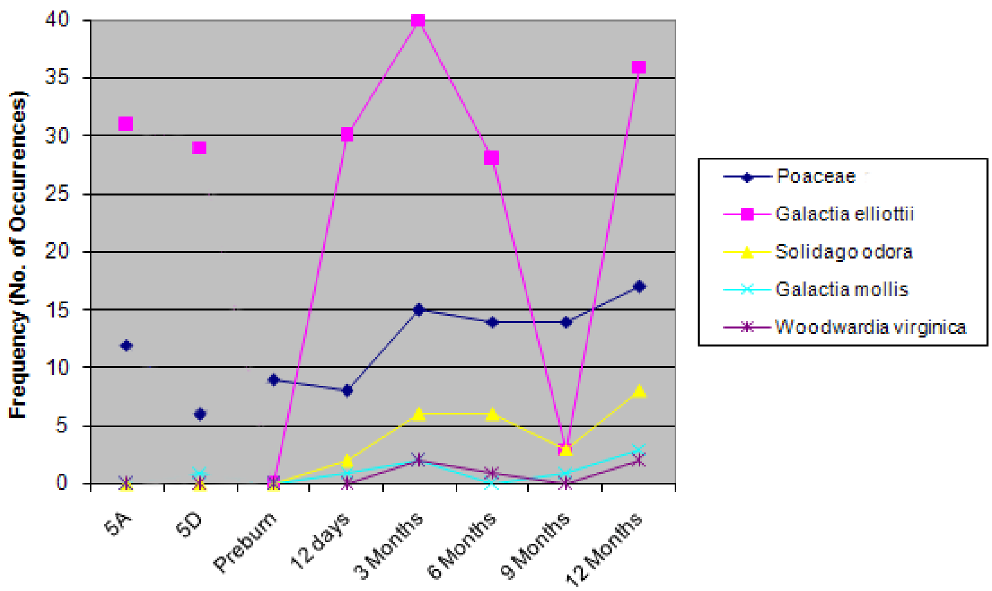


Figure 3-17. Preburn and postburn absolute frequencies of the most abundance herb species in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M. Control values for 5A and 5D are included.

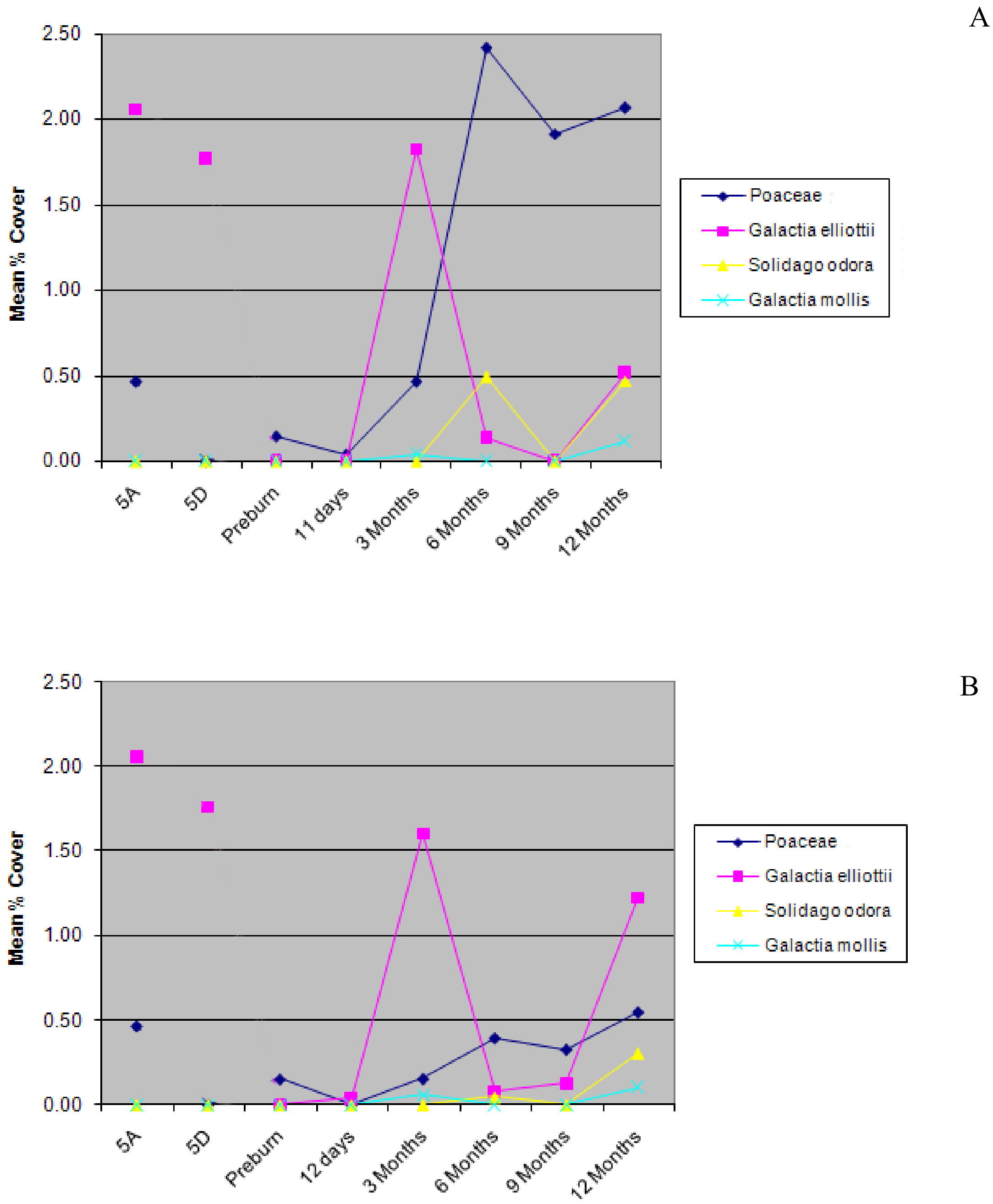


Figure 3-18. Preburn and postburn absolute mean percent cover of the most abundance herb species in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M. Control values for 5A and 5D are included.

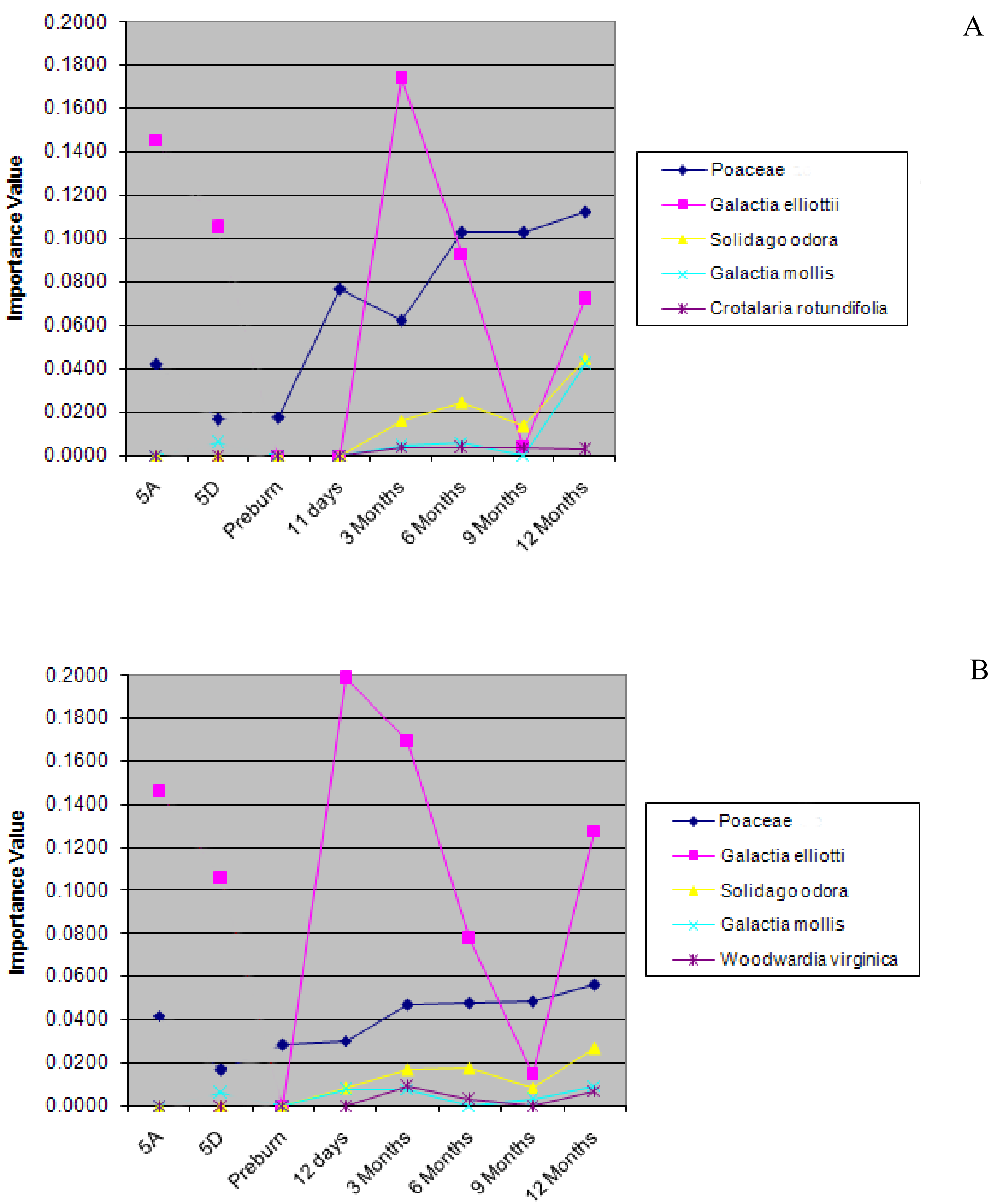


Figure 3-19. Preburn and postburn absolute importance values of the most abundance herb species in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M. Control values for 5A and 5D are included.

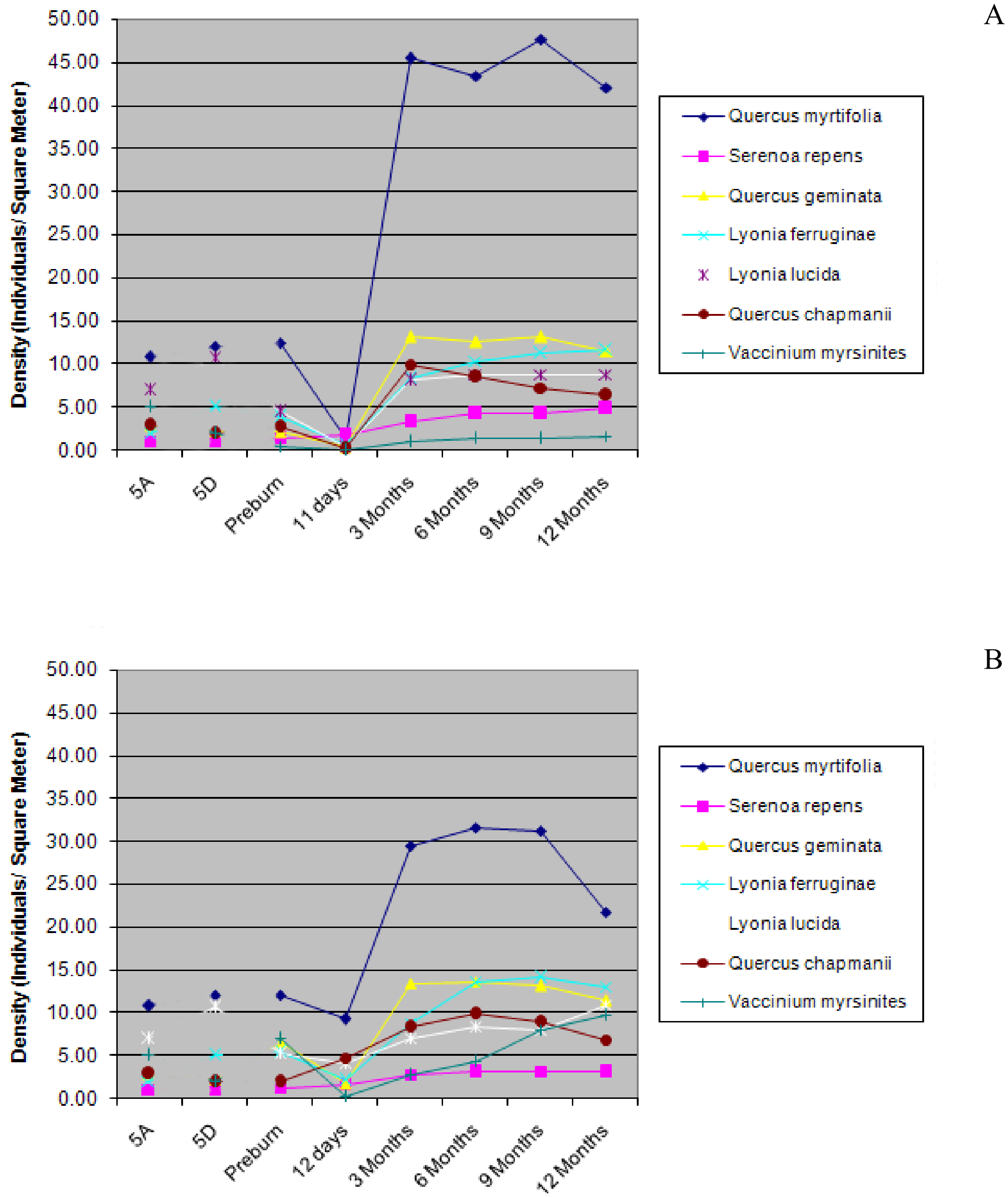
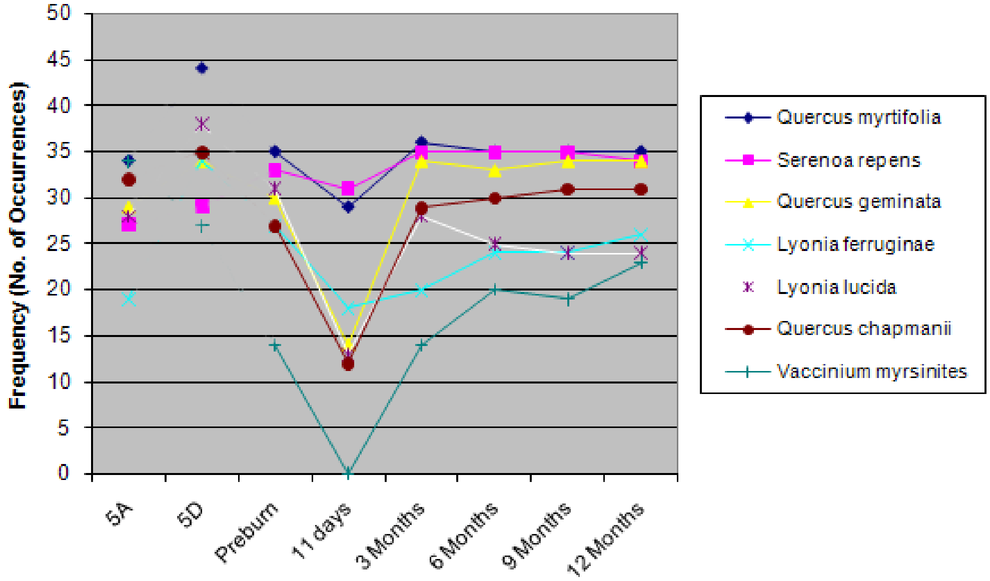


Figure 3-20. Preburn and postburn absolute densities of the most abundance woody species in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M. Control values for 5A and 5D are included.

A



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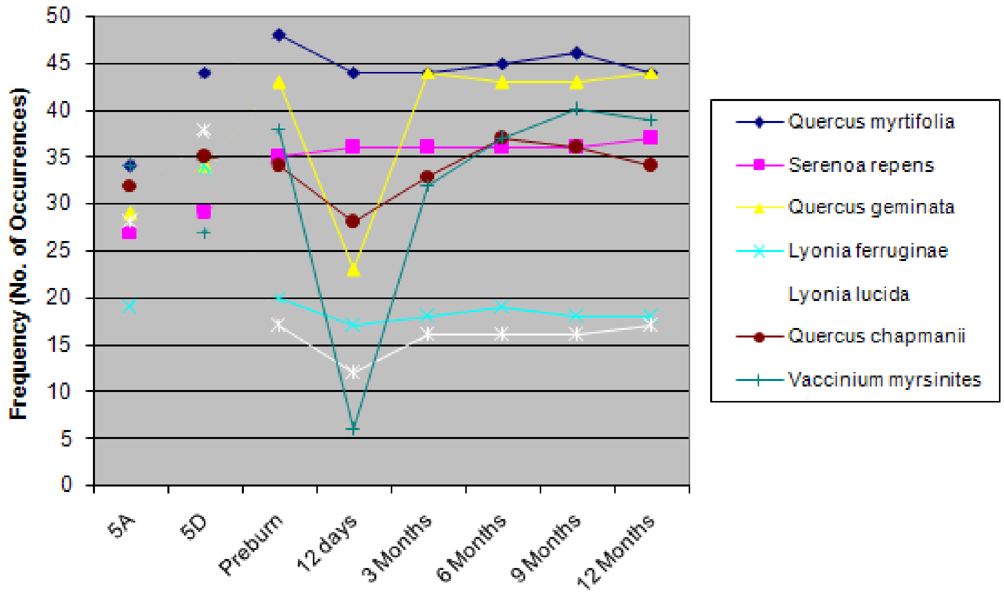
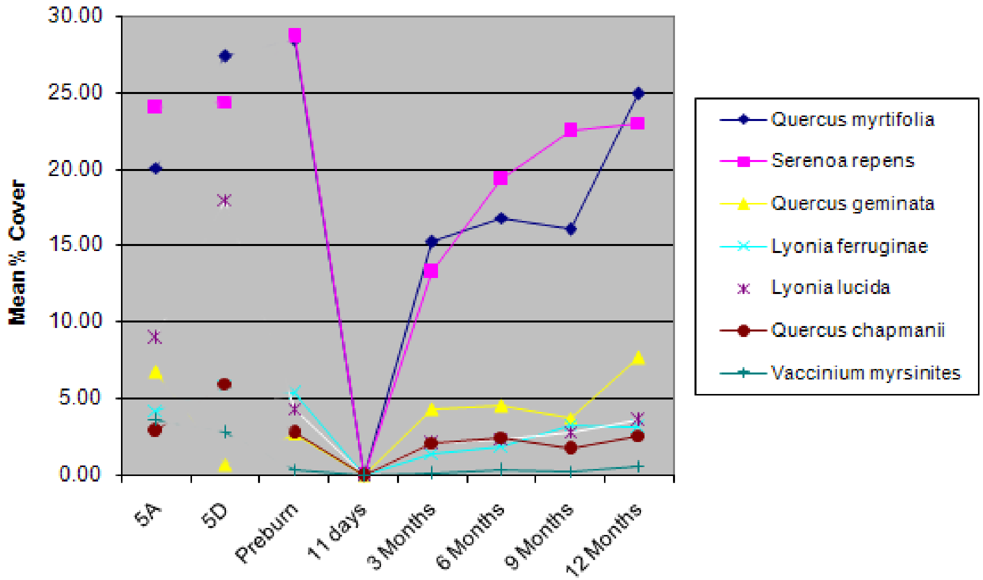


Figure 3-21. Preburn and postburn absolute frequencies of the most abundance woody species in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M. Control values for 5A and 5D are included.

A



B

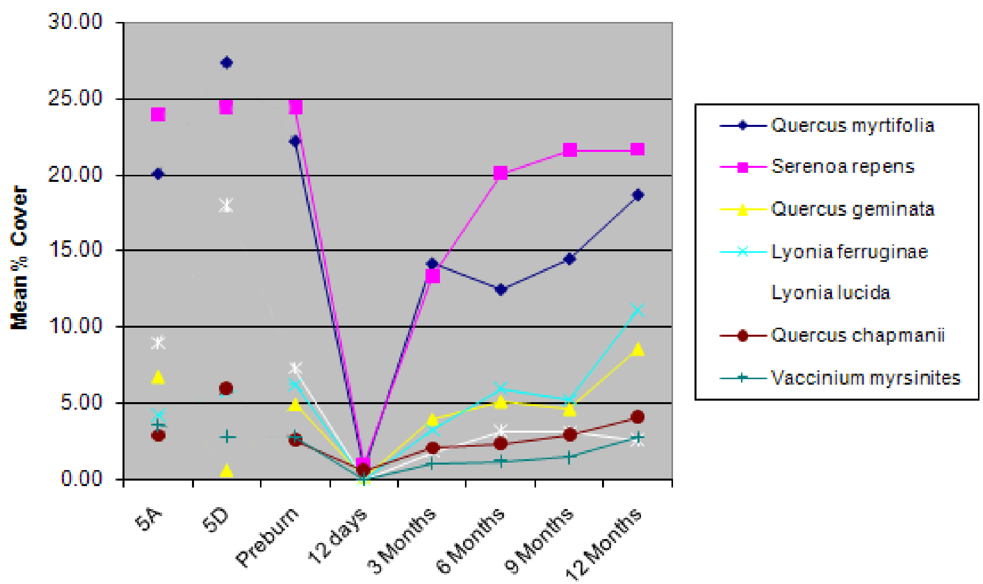
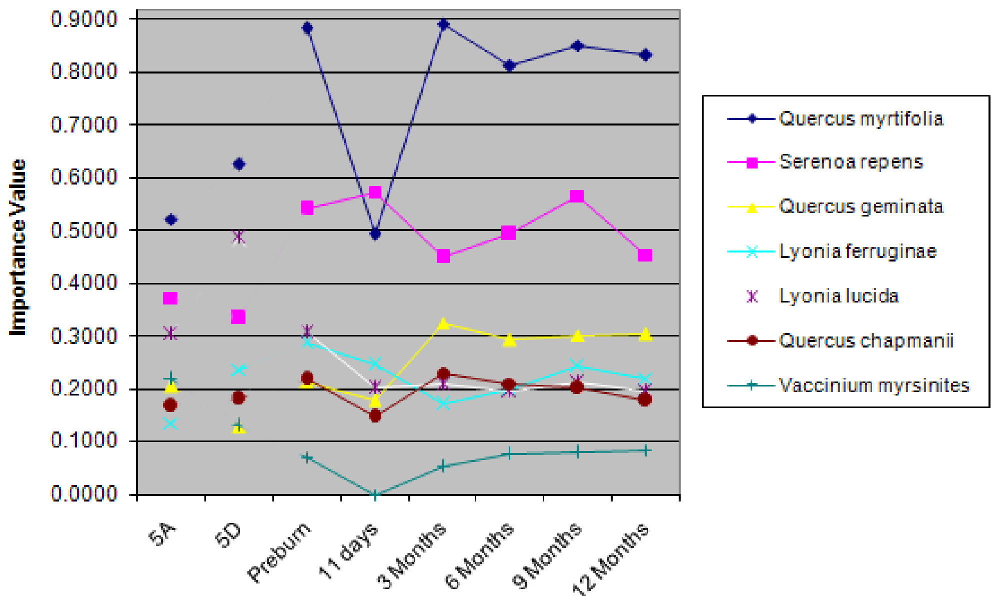


Figure 3-22. Preburn and postburn absolute mean percent cover of the most abundance woody species in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M. Control values for 5A and 5D are included.

A



B

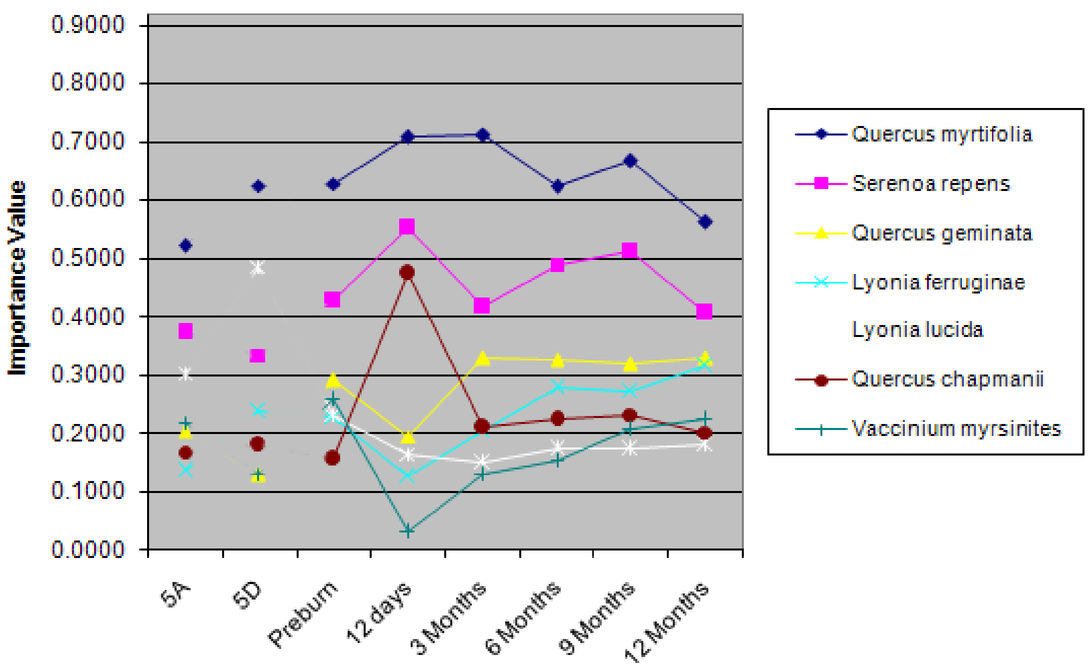
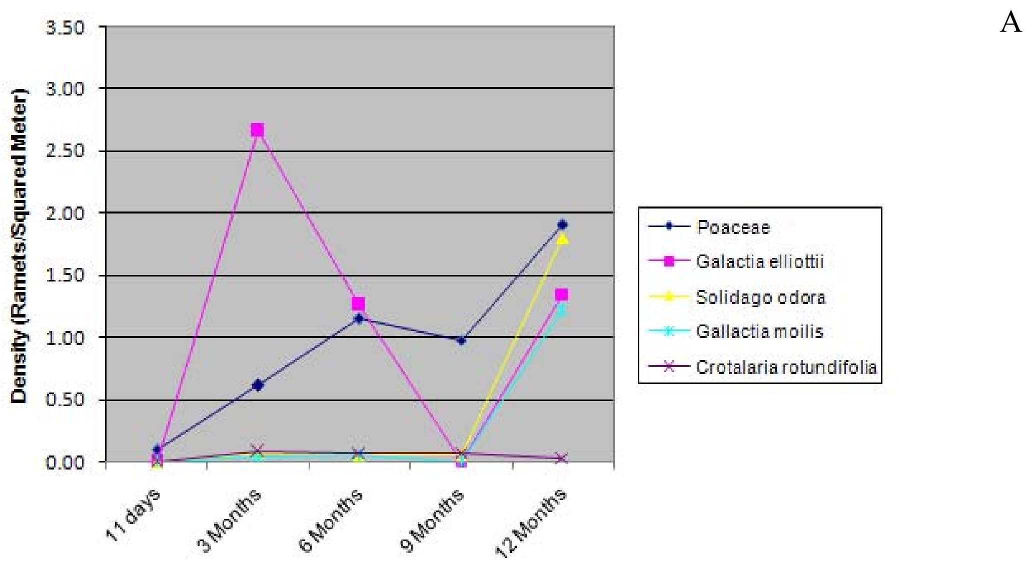


Figure 3-23. Preburn and postburn absolute importance values of the most abundance woody species in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M. Control values for 5A and 5D are included.



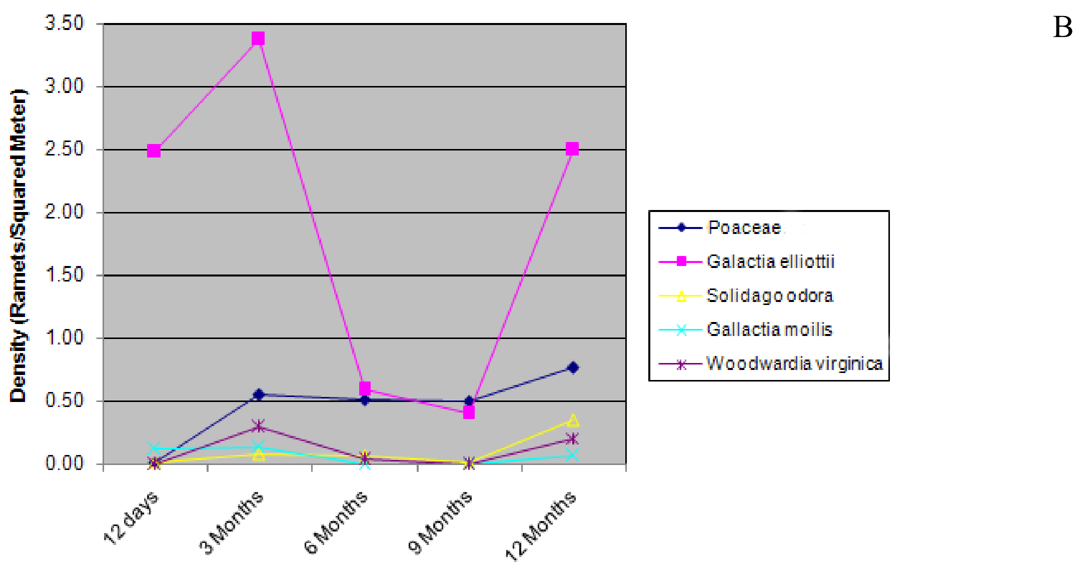


Figure 3-24. Preburn and postburn absolute ramet density of the most abundance herb species in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M.

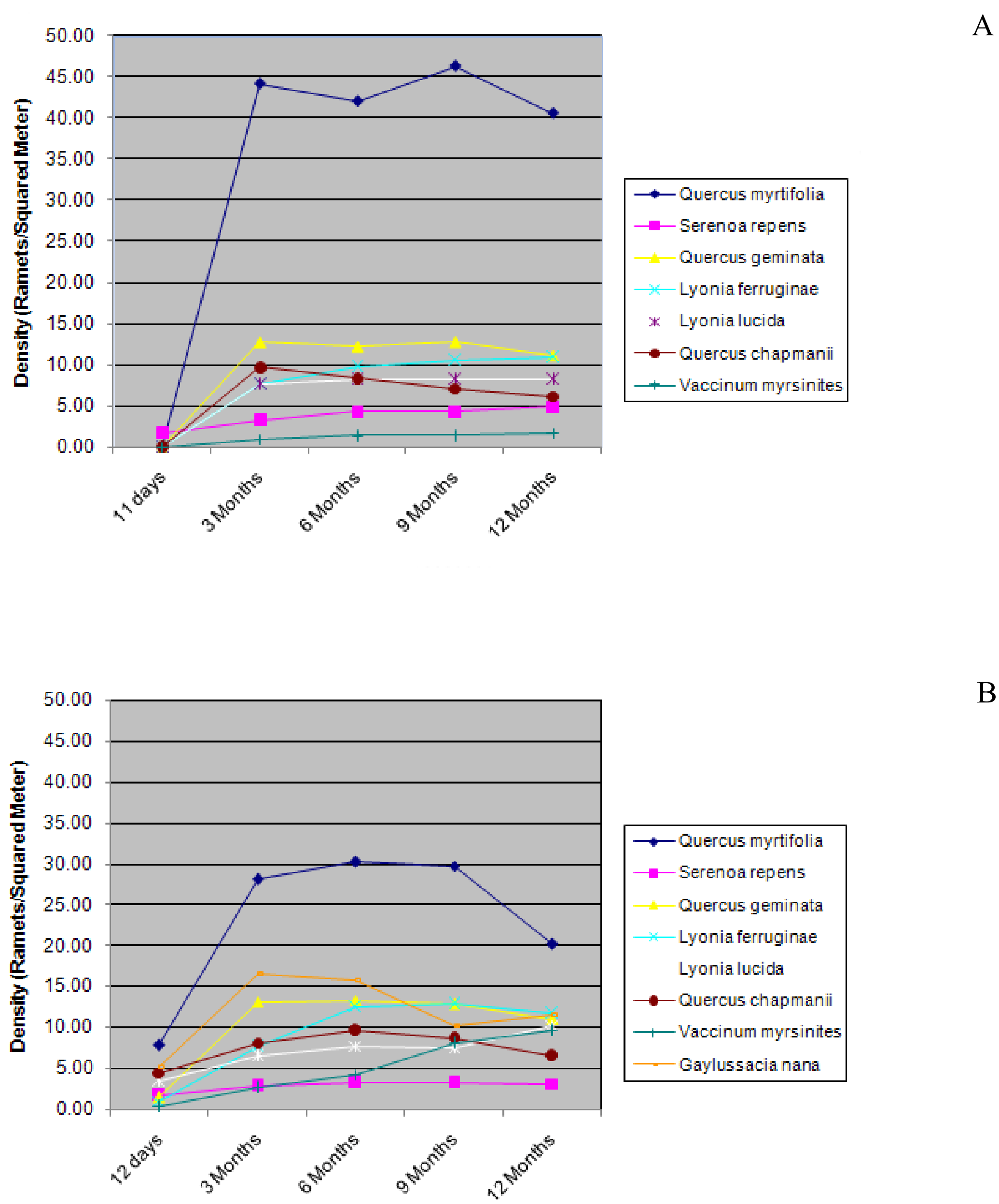


Figure 3-25. Preburn and postburn absolute ramet density of the most abundance woody species in Cedar Key Scrub State Reserve. A) Site 5C. B) Site 2M.

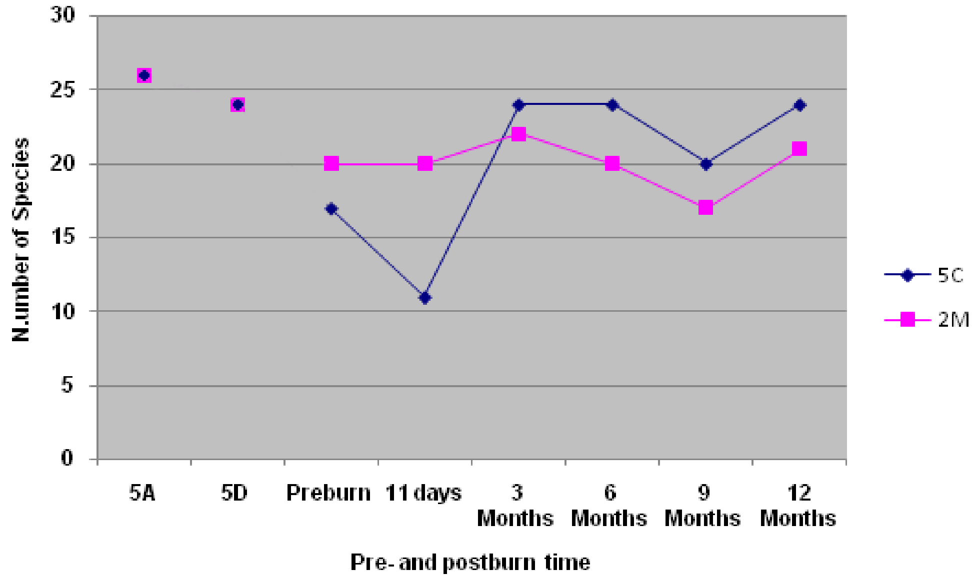


Figure 3-26. Preburn and postburn species richness in treatment sites in Cedar Key Scrub State Reserve. Species richness for control sites 5A and 5D are also displayed.

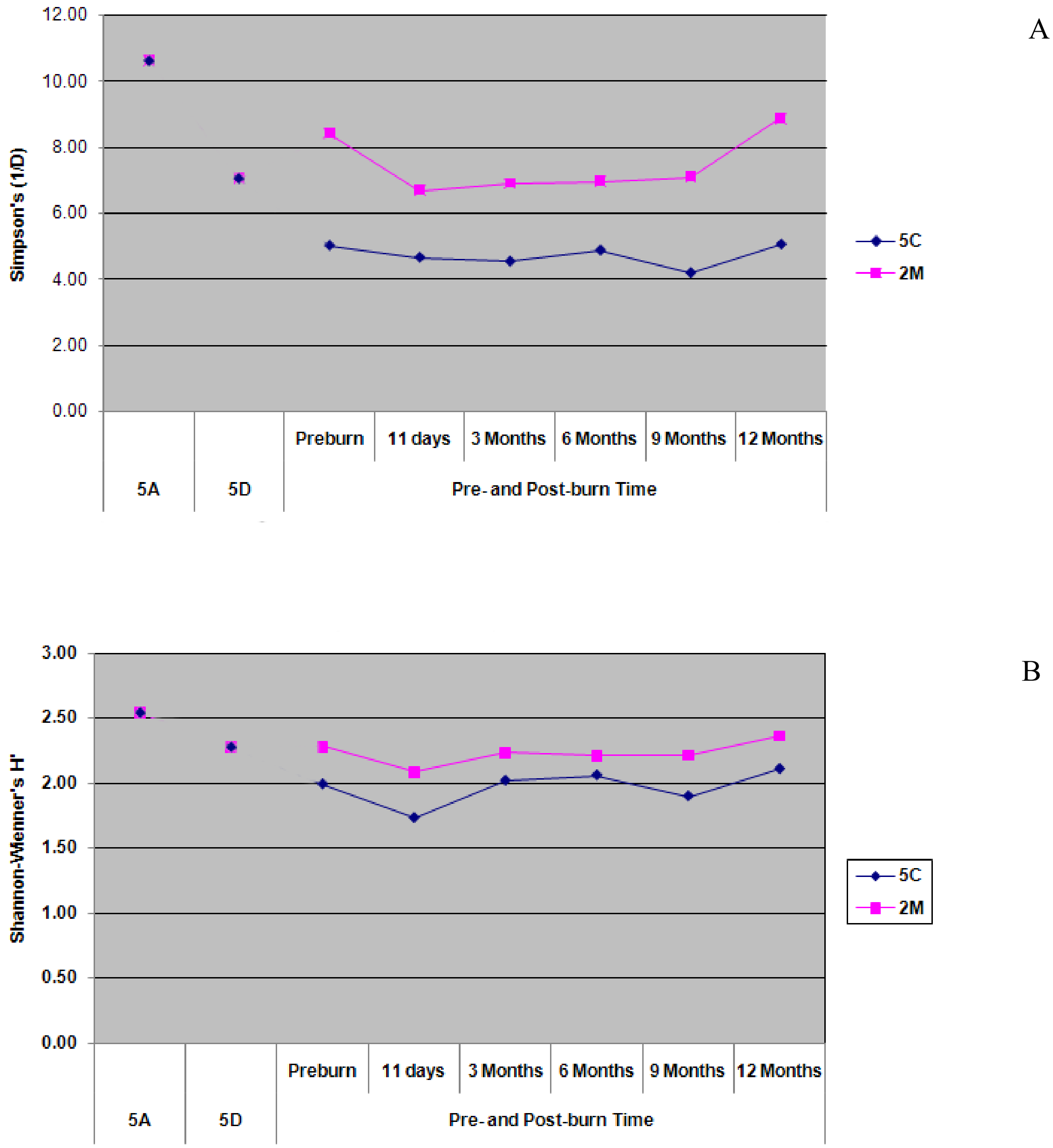


Figure 3-27. Preburn and postburn species diversity in treatment sites in Cedar Key Scrub State Reserve. A) Simpson's index. B) Shannon-Wiener's index. Species diversity for control sites 5A and 5D are also shown.

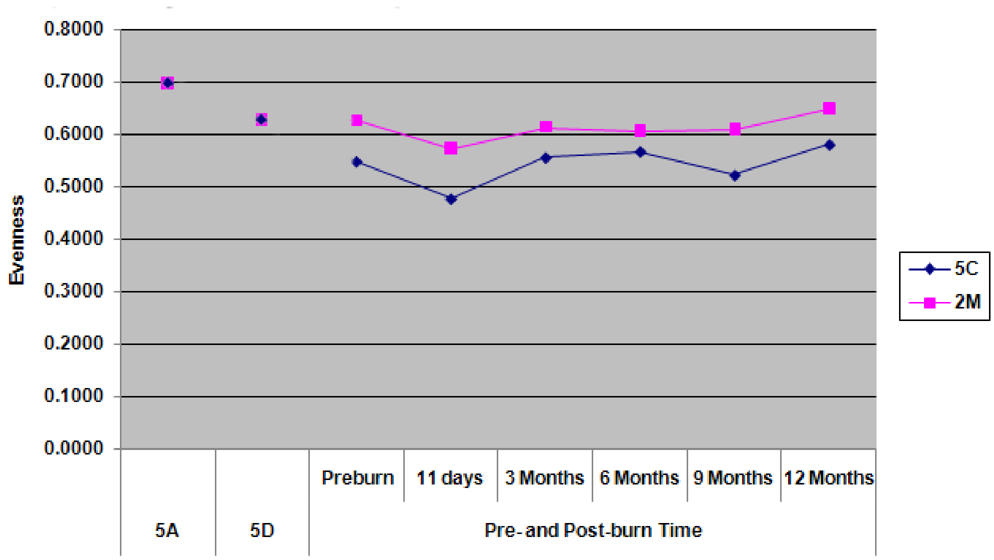
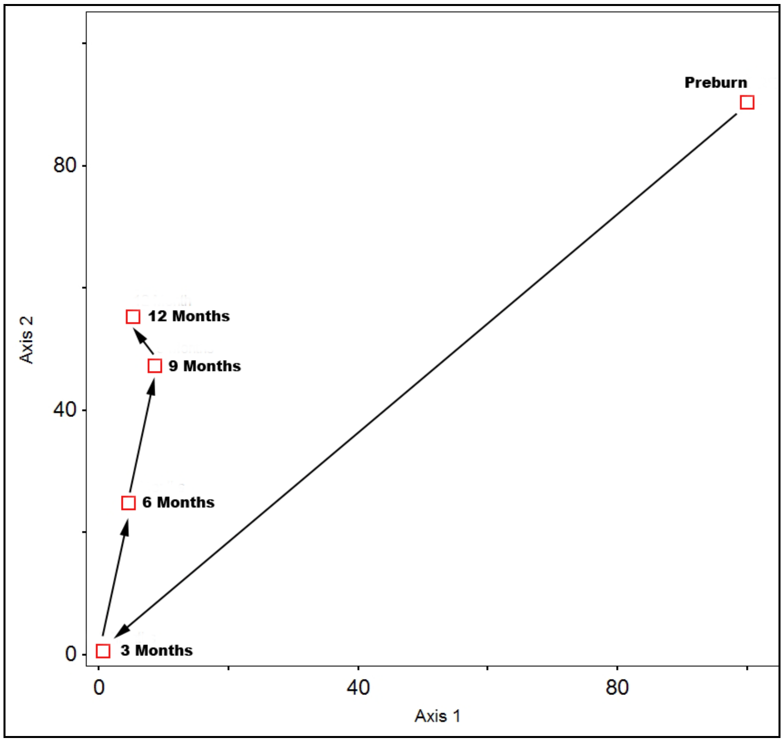
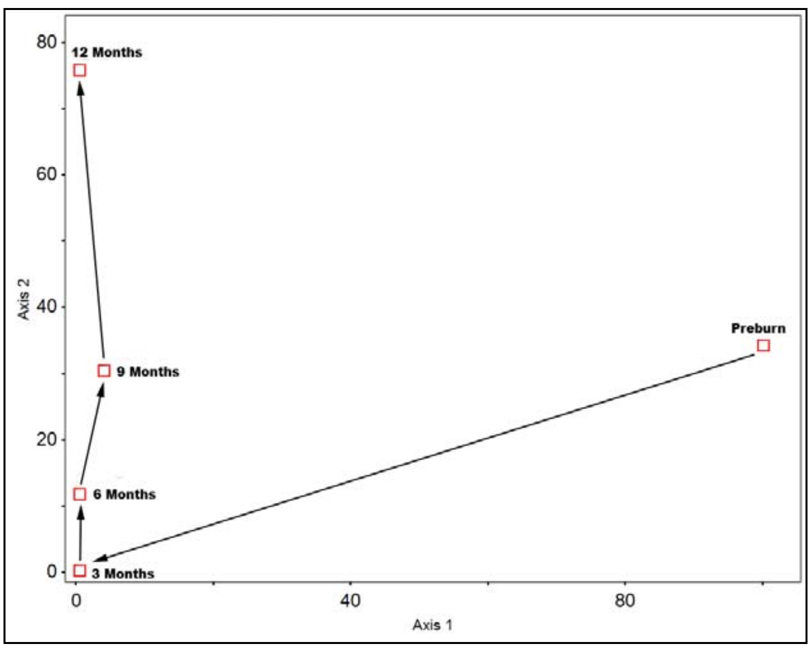


Figure 3-28. Preburn and postburn evenness in treatment sites in Cedar Key Scrub State Reserve. Evenness for control sites 5A and 5D are also displayed.



A



B

Figure 3-29. Detrended correspondence analysis (DCA) sample ordination for densities in 5C in Cedar Key Scrub State Reserve. A) DCA carried out with absolute densities. B) DCA done with relativized densities.

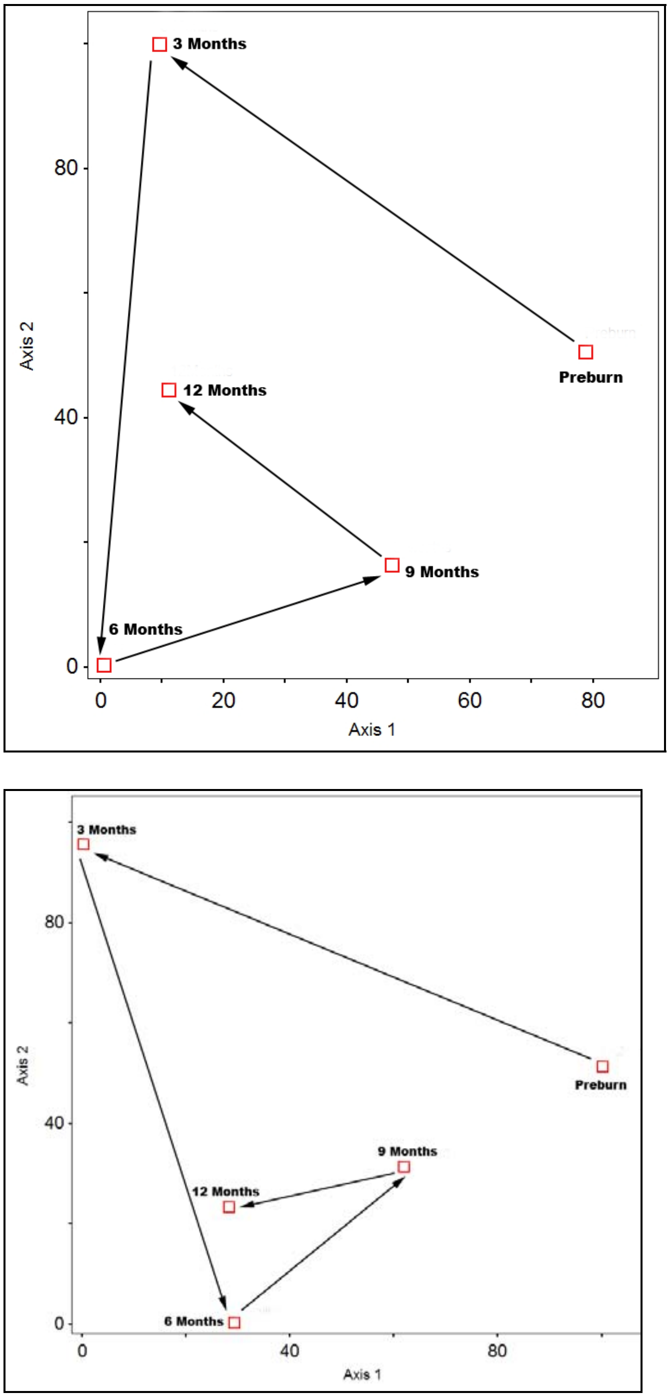


Figure 3-30. Detrended correspondence analysis (DCA) sample ordination for mean % cover in 5C in Cedar Key Scrub State Reserve. A) DCA carried out with absolute mean % cover. B) DCA done with relativized mean % cover.

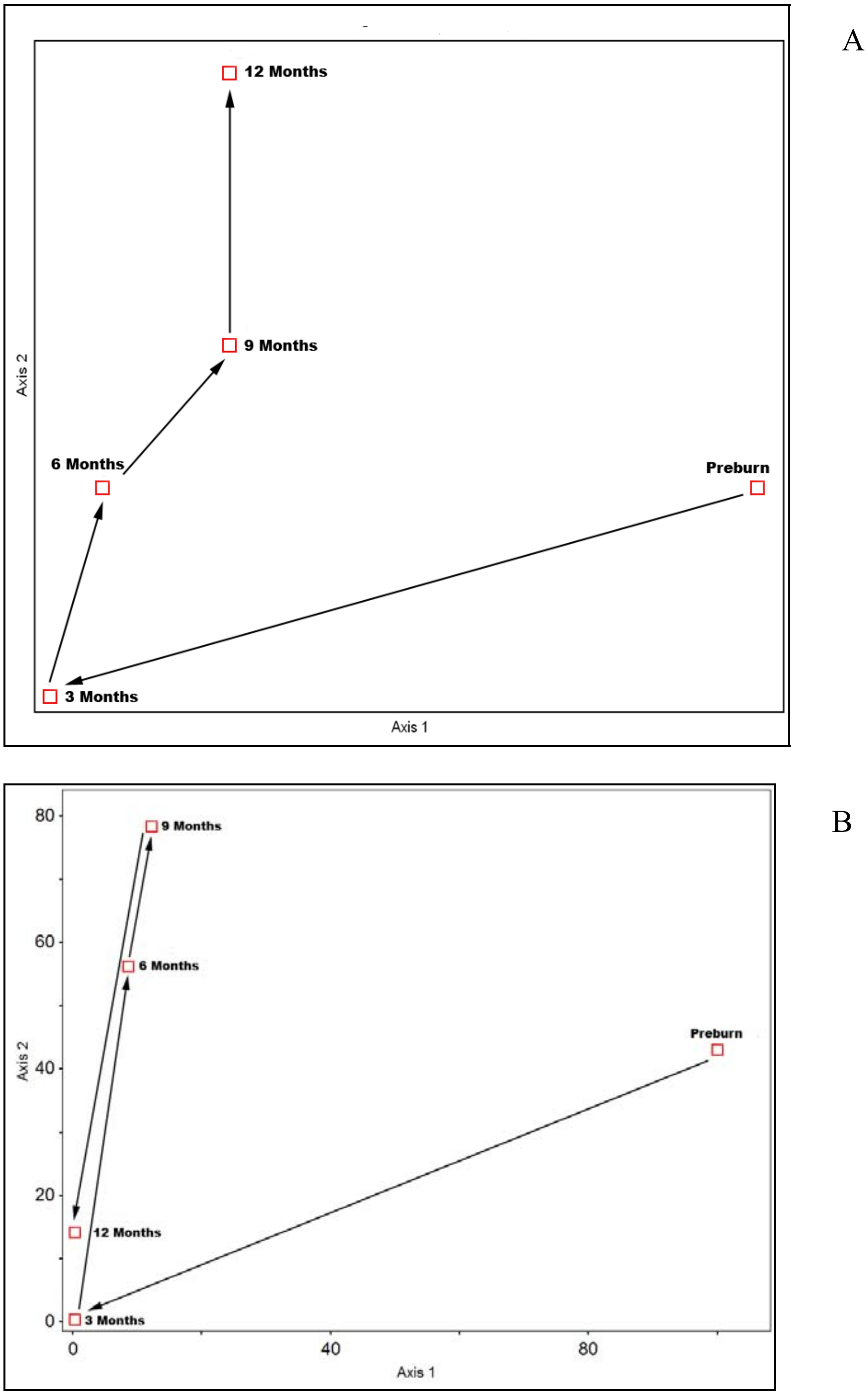
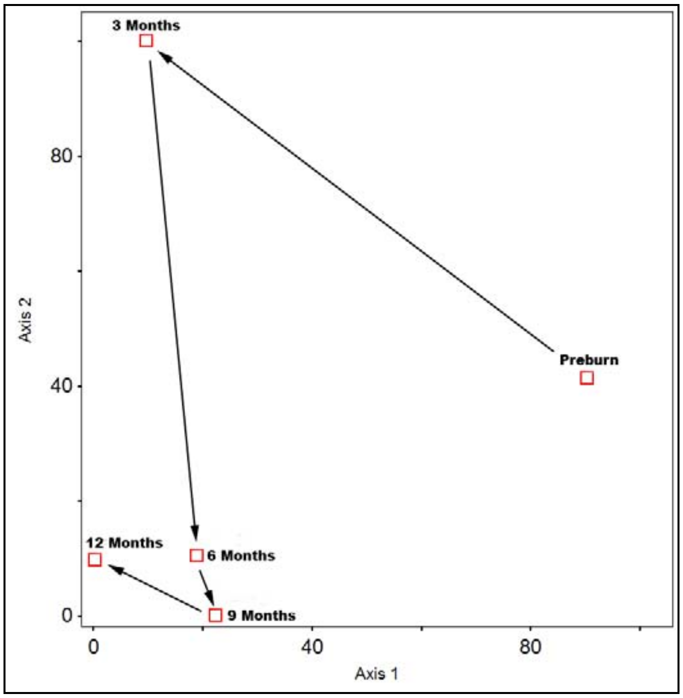
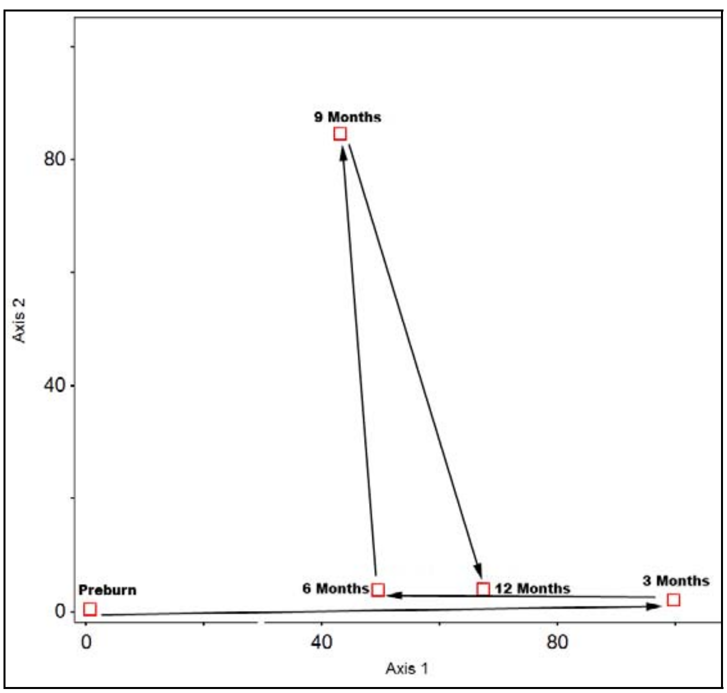


Figure 3-31. Detrended correspondence analysis (DCA) sample ordination for densities in 2M in Cedar Key Scrub State Reserve. A) DCA carried out with absolute density. B) DCA done with relativized density.



A



B

B

Figure 3-32. Detrended correspondence analysis (DCA) sample ordination for mean % cover in 2M in Cedar Key Scrub State Reserve. A) DCA carried out with absolute mean % cover. B) DCA done with relativized mean % cover.

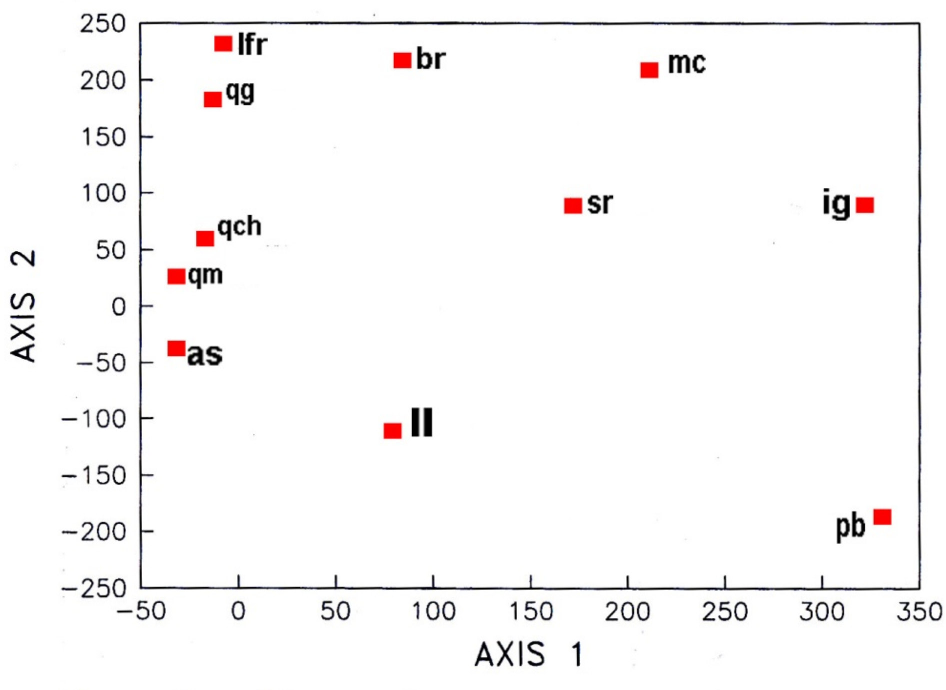


Figure 3-33. Stand and species ordination of oak-saw palmetto scrub based on preburn absolute mean percent cover in Kennedy Space Center. Codes: ll = *Lyonia lucida*, as = *Aristida stricta*, qm = *Quercus myrtifolia*, qch = *Quercus chapmanii*, qg = *Quercus geminata*, lfr = *Lyonia fruticosa*, br = *Brevaria racemosa*, mc = *Myrica cerifera*, sr = *Serenoa repens*, ig = *Ilex glabra*, pb = *Persea borbonia*.

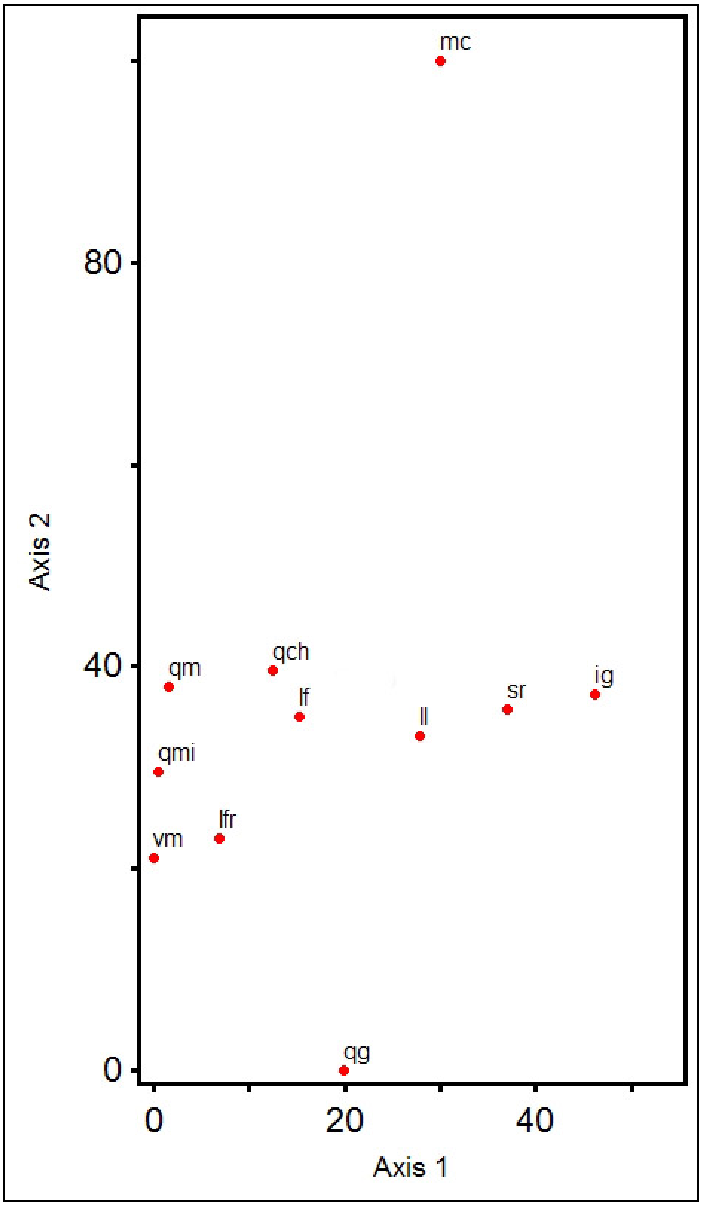


Figure 3-34. Site and species ordination of scrubby flatwoods based on preburn absolute mean percent cover in Cedar Key Scrub State Reserve. Codes: ll = *Lyonia lucida*, lf = *Lyonia ferruginea*, lfr = *Lyonia fruticosa*, qm = *Quercus myrtifolia*, qch = *Quercus chapmanii*, qmi = *Quercus minima*, qg = *Quercus* geminata, vm = *Vacinium* myrsinites, mc = *Myrica cerifera*, sr = *Serenoa repens*, ig = *Ilex glabra*,

CHAPTER 4
SMALL MAMMALS RESPONSES TO PRESCRIBED FIRE

Introduction

Even though prescribed fire is the primary method of fuel reduction in the United States, the effects of controlled burns on fauna are not well understood (Pilliod et al. 2003). Of the five groups of vertebrates, mammals have received more attention. The effects of prescribed fire on amphibians have been summarized by Russell et al. (1999), Bury et al. (2002), and Pilliod et al. (2003). The effects of prescribed burning on birds were reviewed by Smith (2000). General effects of prescribed fire on mammals were summarized by Bendell (1974), Lyon et al. (1978), Wright and Bailey (1982), Peek (1986), and Landers (1987), and general effects of prescribed fire on small mammals were reviewed by Ream (1981) and Smith (2000).

Ream (1981) presented an annotated bibliography of 237 papers, and a very brief summary about prescribed fire effect on *Sorex* spp. (shrews), *Sylvilagus* spp. (rabbits), *Lepus americanus* (snowshoe hare), *Castor canadiensis* (beaver), *Eutamias* spp. (chipmunks), *Spermophilus* spp. (ground squirrels), *Tamiasciurus hudsonicus* (red squirrel), *Glaucomys* spp. (flying squirrel), *Thomomys* spp. (pocket gophers), *Peromyscus maniculatus* (*P. maniculatus*), and *Clethrionomys* spp. and *Microtus* spp (voles).

Smith (2000) presented direct and indirect effects of fire, both wildfire and prescribed burning together in a single discussion, although the majority of the references were prescribed burning studies. He stated that the direct effect of fire (injury or mortality) is lower than the indirect effect through habitat modification. Fires generally do not kill or kill a very small proportion of small mammals because during a fire they use underground refugia, where adequate ventilation is essential for animal survival (Bendell 1974). Immediately after fire, some species leave their habitats and emigrate because of lack of food and cover in the burned area.

Other species immigrate to take advantage of the altered habitat. The length of time before species return depends on how much fire altered the habitat structure and food supply. Thus, each species is likely to respond differently to fire and subsequent habitat changes. Actually, most of the literature about the relationship between fire and small mammals is about how vegetation changes affect their populations. These changes have been mainly reported in terms of abundance and densities following fire. But, little is known about other demographic factors that may be essential for understanding population responses. These are the most general statements about small mammals' responses to fire made by Smith (2000). Another point of view about the effects of prescribed burning is presente below.

Even though the literature presents a variety of responses of small mammals to prescribe fire, there are some aspects that can be synthesized. The majority of the studies cited were of short duration (<30 months) and direct mortality was rarely documented (Tevis 1956, Chew et al. 1959, Taylor 1981, Ver Steeg et al. 1983, Singer and Schullery 1989, Harty et al. 1991); some studies indicated no change in abundance after prescribe burning (Arata 1959, Cook 1959, Writz 1977, Kaufman et al. 1983, Jones 1990, Ford et al. 1999, Vreeland and Tietje 1998); the majority of the studies, however, showed a positive response (population increase) to prescribed fire in almost all habitats (Tevis 1956, Shadowen 1963, Hatchell 1964, Ahlgren 1966, Lawrence 1966, Beck and Vogl 1972, Stout et al. 1971, Kreftin and Ahlgren 1974, Layne 1974, Wirtz 1977, Hon 1981, Kaufman et al. 1982, McGee 1982, Bock and Bock 1978, 1983, Gunther et al. 1983, and Forde et al. 1984, Kaufman et al. 1988a, Wirtz et al. 1988, Jones 1990, Blanchard 1991, Sullivan 1995b, 1995b, Greenberg et al. 2006) with population increases attributed to an increase of abundance in seeds and/or insects (Tevis 1956, Ahlgren 1966, Hooven 1973, Layne 1974, 1990, McGee 1982, Halford 1981, and Gunther et al. 1983). Some studies reported small mammal use of refugia during and after a patchy prescribed burning (MacGee 1982, Schwilk and Keeley 1998); dispersal to unburned areas were described (Tevis 1956, Arata 1959, Lee 1963, Komarek 1965, 1969, Odum et al. 1974, Wirtz 1977, Blankenship 1982, Forde et al. 1984, Wirtz et al. 1988), and dispersal to burned areas was documented in several studies (Cook 1959, Gashwiler 1959, Hatchell 1964, Ahlgren 1966, Sims and Buckner 1973, Krefting and Ahlgren 1974, Layne 1974, Schramm 1983, Martell 1984, Vacanti and Geluso 1985, Kaufman et al. 1988b, 1990, Monroe and Converse 2006). Differences in fire regime and management practices (fire in combination with logging, chopping, clearcutting, and mowing), habitat types, topography, and climate among studies make generalizations difficult to draw. However, there are two aspects about the effects of prescribed burning that are relevant because of their direct connotations for researchers and land managers. These aspects are the use of surrounding un-burned habitats as refugia and the influence of the re-growth of the vegetation on re-colonization.

Few studies have indicated that adjacent unburned habitats might be used as refugia and population recovery is due to plant species regrowth several months/years after prescribed fire. Goatcher (1990) and Blanchard (1991) carried out research of the possible use of stream-terrace hardwood forest as refuge for *Peromyscus gossypinus* in Lee Memorial Forest, Baton Rouge, Louisiana. They used live-trap capture, radio telemetry, and fluorescent tracking pigments to monitor movements before, during, and after prescribed burning. No movements across the fire-break were detected by these methods. They concluded that *P. gossypinus* apparently does not use stream-terrace hardwood forest as refuge after prescribed fire in adjacent pine forests. However, more research is needed because several studies have documented that small mammals temporarily leave burned sites. If they temporarily abandon burned sites, most likely they use other habitats as temporary refugia. Also, the following studies have indicated that small mammals recolonize burned sites after the regrowth of the vegetation. *P. maniculatus* and *Spermophilus armatus* (Uinta ground squirrel) populations approached control numbers after three years following a spring burning, when total cover of the understory was near control levels in Burro Hill, Bridger-Teton National Park, Wyoming (McGee 1982). The impact of fire on small mammal communities in the central Appalachians in Pennsylvania was transitory, and the differences in small mammal abundance between unburned and burned sites disappeared within eight months after fire. This rapid recovery of small mammal populations was explained by the fast regrowth of ground cover within the study area (Kirkland et al. 1996). This link between small mammals and regrowth of the vegetation was also used as explanation for population recovery in the study conducted by Ahlgren (1966) in Minnesota and Sullivan and Boateng (1996) in British Columbia. Both studies found an increase in *P. maniculatus* on burn sites following fire and a decrease in *Clethrionomys gapperi* (southern red-backed vole) numbers 2-3 years following fire until recovery of the ground cover vegetation occurred. *Neotoma mexicana* (Mexican woodrats) benefits from thinning and/or prescribed burning that encourages shrub densities in the long-term by reducing canopy cover in Coconino National Forest, Arizona (Converse et al. 2006). Research on recovery of small mammal populations relative to development rate of vegetation structure following fire is highly needed (Taylor 1981). In Florida, unfortunately, few studies have been conducted to provide evidence of the impact of prescribed fire on small mammals, and no study has ever evaluated the importance of adjacent habitats to burned sites as refugia. In addition, little documentation has been reported on how closely population recovery is linked with the regrowth of the vegetation.

Only 11 studies have evaluated the effects of fire (nine prescribed burning and two wildfires studies) on small mammals in Florida. Arata (1959) found no change in the composition of the populations of *P. floridanus*, *Peromyscus polionotus* (old-field mouse), and *Sigmodon hispidus* before and after burning in longleaf/turkey oak habitat in north-central Florida. Also, he reported that *S. hispidus* moved from burned to unburned areas, while *P. polionotus* and *P. floridanus* stayed in the burned areas. Therefore, prescribed fire did not have a detrimental effect on these species. Even though Vogl (1973) trapped only during five nights and obtained a low trapping success, he stated that the densities of *Peromyscus gossypinus*, *S. hispidus*, and *Blarina brevicauda* (short tailed shrew) appeared to be similar in the burned and unburned hardwood forest in north Florida (Gannet Pond, Leon County). Gates and Tanner (1988) found no apparent effect of prescribed burning on *Geomys pinetis* (pocket gopher) at the successional stages of sandhill communities in Ordway-Swisher Preserve (OSP, Putnam County) in north-central Florida. Jones (1989, 1990) found that three populations of *Podomys floridanus* had little or no mortality due to prescribed fire, and populations were higher on burned areas that on unburned sites in longleaf/turkey oak habitat in OSP. In addition, she found that all individuals except one did not move out of the burned area after prescribed burning. Layne (1974) reported a population increase in *P. gossypinus* and *S. hispidus* after a wildfire in slash/longleaf pine habitat in north-central Florida and suggested that burned areas could act as "dispersal sinks." Layne further stated that the reappearance of *Reithrodontomys humulis* (Eastern harvest mouse) and *S. hispidus* on the burned area appeared to be correlated with redevelopment of the ground cover. Layne (1990) also conducted a long-term monitoring of *P. floridanus* population in sand pine scrub at Cedar Key Scrub State Reserve (CKSSR), Levy County. He found that the species survived a heavy wildfire in 1955 and was still present in 1986; however, absolute density and relative abundance declined 10 years after fire. Layne (1990) also reported that *P. floridanus* was still present at low numbers in sandhill and scrub sites that were burned in 1927 in Archbold Biological Station, Highlands County. In comparison, populations were higher and more stable in similar nearby habitats that were burned periodically. According to Layne (1992), *P. floridanus* populations are higher in early successional stages of scrub and sandhill vegetation following fire. Later, in the absence of fire, populations decline as habitat structure becomes denser, shadier, and microclimatic conditions more mesic. Even though Layne's papers (1974, 1990) reported on the effect of wildfires, these papers are included in this introduction because of the low number of studies on the subject in Florida. Fitzgerald (1990) stated no significant treatment effects were detected on *S. hispidus* and *P. gossypinus* in dry prairie of Myakka River State Park in southwestern Florida. Jones (1992) reviewed 38 papers on the effects of fire on *Peromyscus* and *Podomys*, and she found that the majority of the papers described responses of *P. maniculatus*, whose numbers increased on burned areas in forest and grasslands. Other species differed in their response to fire according to the type of habitats. *P. floridanus* appeared to have little or no short term effects following prescribed burning, and abundance equaled or exceeded pre-fire levels after two or three months. Depue (2005) found that *P. floridanus* in central Florida increased or recovered to pre-burn levels within six months following prescribed burning in Bullfrog Creek Mitigation Park; it dropped in numbers following prescribed fire, but started to increase when the study ended in Split Oak Mitigation Park; and the decrease in animal numbers remained unaffected by prescribed fire in Chuluota Wilderness Area. No study has ever determined the response of *Ochrotomys nuttalli* to prescribed fire in Florida.

After reviewing the literature on the effects of prescribed burning on small mammals in the United States, I found that the majority of the studies had methodological problems that did not permit comparison among them and the possibility of making generalizations. A high proportion of these studies were short-term (from weeks to one year) and limited in geographic area, fire behavior characteristics were not measured, they had small sample sizes, and lacked pre-fire data, experimental control, replication, and randomization. Therefore, these studies are neither true experimental designs nor quasi-experimental ones that could provide strong causal inference and domain (James and MacCulloch 1995). According to Pyne et al. (1996), two methods have been frequently used in fire studies. The most common one is to compare burned with unburned (controls) areas, but two important assumptions are not validated. First, treatment and control must be ecologically similar. Hence, soils, slopes, species composition, vegetation structure, and fire history need to be similar. Second, fire behavior characteristics must not vary between treatments. The second common method of study is pre-burn versus post-burn comparisons. In this type of study, only the second assumption needs to be made and validated. However, this approach also needs control, replications, and randomization when it is possible. Whelan (1995) and Russel et al. (1999) recommended that future prescribed fire studies should have more rigorous experimental designs, including larger sample sizes, pre-fire baseline data, more carefully selected controls, and better replications. Another aspect of importance among prescribed fire studies is the way data analysis has been conducted.

The majority of the studies on prescribed burning effects on small mammals focus at the population level, providing information about changes in abundance indices and densities after fire. The problem with evaluating fire effects by using abundance indices, such as minimum number alive or catch per unit effort, is that these indices tend to be biased in their estimates of the true abundance. They are also biased because they do not account for differences in detection probabilities that are likely in small mammal studies. These include differences in detection probabilities between individual animals over time, or in response to an experimental treatment

(Monroe and Converse 2006). In contrast, few studies (Lee and Tietje 2005, Converse et al. 2006, Monroe and Converse 2006) have used other demographic factors such as survival and recapture probabilities to analyze prescribed fire effects. This approach might be critical for a better understanding of prescribed burning effects.

Since prescribed fire has been extensively used in Florida as a management tool for restoring fire-adapted ecosystems (i.e., longleaf pine-sandhill, sand pine scrub, and scrubby flatwoods), prescribed burning effects on small mammals become very important because of their roles in ecosystem functions. Small mammals constitute the prey for many forest predators (Zeilinski et al 1983; Williams et al. 1992). They influence the structure of vegetative communities through seed predation and dispersal (Vander Wall, 1993; Hollander and Vander Wall 2004; Schnurr et al. 2004). They play an essential role as dispersers of ectomycorrhizal fungi (Pyare and Longland 2001). Therefore, more studies on prescribed burning effects are needed because several small mammal species have restricted geographical ranges, occurs in only localized habitats that may be vulnerable to management practices, or may be listed under the Endangered Species Act. In addition, no study has ever assessed the role of adjacent habitats as refugia, and no study has been conducted to evaluate the immediate (days after burning) and short-term effects (one month after initiation of plant species growth, 6 months, and one year after burning) of prescribed fire on small mammal species in Florida. Furthermore, no information is available on survival rate when small mammals leave the burned area. These aspects may be critical for population survival and have been overlooked in fire studies. These topics of research are relevant for fire managers and those responsible for assessing the potential effects of prescribed burning on rare, sensitive, and endangered small mammal species in Florida. Managers, researchers, and non-game species and their habitats will all benefit from a more comprehensive understanding of how small mammal species respond to prescribed burning.

Objectives and Research Hypotheses

The objectives of this study were to document small mammal responses to prescribed fire, to evaluate the importance of vegetation surrounding wetlands next to burned sites as refugia, and to determine whether or not population recovery is linked with the re-growth of the vegetation. The research hypotheses were the following:

1. *P. floridanus*, *S. hispidus*, *P. gossypinus*, and *O. nuttalli* in Cedar Key Scrub State Reserve use the vegetation surrounding wetlands next to burned sites as temporary refugia after prescribed burning.
2. Prescribed burning does not have a negative effect on the survival probability of *P. floridanus* and *S. hispidus* in Cedar Key Scrub State Reserve.

Methodology

Trapping Methods

Four 10x10 grids with 10 trap lines were used for capturing, marking, and recapturing mice. Grids were installed in the scrubby flatwoods of treatment and control sites. Each grid had 100 standard-sized Sherman Live Traps (7.6 cm x 8.9 cm x 22.9 cm) arranged in ten lines with 10 trapping stations each and 15 m between trapping stations. Each trapping station was identified by a flag. Also, two trap lines with 10 traps each (15 m between traps) were placed between each grid and the wetland next to it. Trap lines were in the vegetation near the border of each wetland. The idea was to detect movements between the grids and trap lines. In addition, two traps were located at the entrance of each burrow found in grids and trap lines, and 4-6 traps were placed around the small wetlands found inside treatment and control sites. Each trap was baited with a 50-50 mix of crimped oats and sunflower seeds, and polyester was used as nesting material during periods of cool weather. Palmetto fronds were used to shade and insulate traps. Traps were checked at sunrise and mid afternoon because of the diurnal activity of *S. hispidus*, and traps were left set at all times during the session.

Four trapping sessions were conducted before and after prescribed burning (Figure 4-1). During the eight trapping sessions, trapping was performed up to a 5-day sequence to avoid possible loss in body mass associated with capture (Slade 1991). The 3rd trapping session was carried out right after three hurricanes hit Cedar Key in August-September 2004, and the 5th trapping session was done after prescribed burning. In treatment and control sites, pre-treatment data were collected from 03/02/04 to 04/20/05, and post-treatment data were collected from 04/26/05 to 07/19/06. Collection of post-treatment data started 5 days after prescribed burning and continued at intervals of 3 months after that. In 2006, the 8th trapping session in February, just 9 months after burning, was cancelled because of the low temperature and rescheduled for April 2006.

Trapping effort was increased at three opportunities. The first one took place right after the 3rd trapping session, in which five trap lines (10 traps each; 15 m between traps) were installed on places of higher elevation in/near each treatment and control sites during five nights. The purpose of this trapping was to recapture 79 individuals marked during the 1st and 2nd trapping sessions and not recaptured during the 3rd session. The second effort occurred after prescribed burning, and it was included during the 5th trapping session and in treatment sites only. A trap line (10 traps) was placed at the east and west side of the grid in 5C and at the east and south side of the grid in 2M. Also, 18 traps were added to the two trap lines located in the vegetation surrounding the wetland in 5C and 2M. The idea was to detect movements of marked individuals from the scrubby flatwoods to wetlands after burning. The third effort was carried out only in control sites 5D and 5A in June and July 2005, respectively, because these sites were mowed with the exception of the grid and 150-200 m buffer zone around the grids. Trapping occurred in half a grid for five nights to determine if mice left the site because of the mechanic activities near the grid in each site.

Collected data included species, weight, sex, reproductive condition, trap location, and tag number. Reproductive conditions were as follows: juveniles with varying stages of gray pelage, subadult (non-breeding) individuals with adult pelage but not evidence of current sexual activity, and breeding animals with adult pelage and evidence of sexual activity. Sexual activity was determined in males by testicles in scrotal position and for females by pregnant conditions and nipple size. Each mouse was identified with two unique ear-tags (National Band and Tag Co., Covington, KY) and released at point of capture.

Trapping for predators was carried out to remove them from the grids before the eighth trapping session in each site. *Procyon lotor* (raccoon), *Urocyon cinereoargenteus* (grey fox), *Didelphis marsupialis* (opossum), and *Mustela frenata peninsulae* (Florida long-tailed weasel) were the most problematic predators found in treatment and control sites. From the 1st to the 7th trapping session, predators disturbed traps for not more than 3 days. The strategy was to close traps and wait 3-5 days for predators to move to other places. However, before the eighth trapping session, predators were abundant (particularly *P. lotor*) in treatment sites. Therefore, trapping predators took place for 7-10 days in each site before trapping small mammals. Predators were released at a minimum distance of 3.0 km from the grid of capture.

Data Analysis

Due to the small sample size for the capture-recapture dataset, data analysis was only carried out for *P. floridanus* and *S. hispidus*. In addition, the data for treatment sites were combined for each species, and the same procedure was done for control sites. Data collected during the extra trapping effort done only in control sites in June-July 2005 were not considered for the analysis. I evaluated the effect of prescribed burning on the survival probability of *P. floridanus* and *S. hispidus* by using information-theoretic model selection and inference framework (Burnham and Anderson 2002). Also, I used the program MARK 4.3 (White and Burnham 1999) for testing lack of fit and for estimating the survival and recapture probabilities by using the Cormack-Jolly-Seber model (Cormack 1964, Jolly 1965, Seber 1965).

Assumptions of the Cormack-Jolly-Seber model

The Cormack-Jolly-Seber (CJS) model is used alone to estimate survival and recapture probabilities. This model requires information only on the recapture of the marked animals, and that these individuals are representative of the populations. Several assumptions have to be made to be able to estimate the parameters associated with this model. These assumptions are as follows (Williams et al. 2002):

1. Every marked individual at time (i) has the same probability of recapture (pi).
2. Every marked individual immediately after time (i) has the same probability of survival to time (i+1).
3. Marks are not lost or missed.
4. All samples are taken in a short period, and captured animals are released immediately.
5. All emigration from the sampled area is permanent.
6. The fate of each animal regarding capture and survival probabilities is independent of the fate of any other animal.

The goodness of fit (GOF) test

The GOF test was used to test the lack of fit of the data to the underlying assumptions of the CJS model. The survival probability phi was considered constant (**.**), time dependent (**t**), group dependent (**g**; treatment – control), and dependent of the interaction group and time (**g*t**). The recapture probability was also considered under the same conditions, and the combination of all these possibilities added up 16 models. This set of models was used as the candidate model set. I fitted these models to the data by using the “pre-defined model” option in MARK and carried out the GOF test for the full time-dependent model phi(g*t) p(g*t). This model is the general model because it contains the largest number of parameters. The idea was to assess if this model adequately fit the data, which means verifying whether or not the arrangement of the data meet the expectation determined by the assumptions underlying the CJS model.

The GOF test was done by using Bootstrap and Release methods. Both methods estimate the variance inflation factor (c-hat), which quantify the lack of fit of the model to the data or the amount of under or over dispersion that we have in the dataset. The Bootstrap method provides two ways to estimate c-hat: (a) the deviance method which divides the observed deviance (obtained from the summary statistic of the general model) by the mean deviance from the bootstrap summary statistics, and (b) the c-hat method that divides the observed c-hat (obtained from the summary statistic of the general model) by the mean c-hat from bootstrap summary statistic. The Release method presents three tests of which Test 2 and Test 3 check if the 1st and 2nd assumptions are met. I assumed that the 3rd, 4th, 5th, and 6th assumptions were met. C-hat was estimated by dividing the overall chi-square from Test 2 and Test 3 by the overall degree of freedom.

I followed Cooch and White’s (2006) recommendations regarding which c-hat to choose between Bootstrap and Release. These authors suggest to choose the largest c-hat value in the interval 1 < c-hat < 3 in order to make the model selection more conservative and minimizing the chances of Type II error. A c-hat value in the interval 1 < c-hat < 3 means overdispersion of the dataset, and an adjustment for lack of fit is needed with MARK by using that particular c-hat value. So, MARK displays quasi-likelihood Akaike’s Information Criterion values (QAIC) after adjustment. A c-hat > 3 means that the general model does not fit the data. A c-hat = 1 means a perfect fit and the data do not need any adjustment. A c-hat < 1 means underdispersion of the data, and Cooch and White suggest to treat a c-hat < 1 as a c-hat = 1. These criteria allowed choosing a c-hat value for the general model and made the corresponding correction to it and the candidate model set.

Model comparison, model selection, and hypothesis testing

To compare and select models, the following steps were carried out. First, selecting the most parsimonious model in the candidate model set by using the Akaike's Information Criterion (AIC). The most parsimonious model is the one that best explains the variation in the data with the lowest number of parameters, and it is better supported by the data. AIC is a good and well-justified criterion for selecting the most parsimonious model and it is considered a robust way of model selection (Burnham and Anderson 2002). The criteria for model selection were the following: AIC, Delta AIC, the normalized Akaike weights, model likelihood, the number of parameters, and model deviance.

The AIC index measured how much the model explained the variation in the data. The AIC = $-2\ln(£(q/data))+2K$ = -2 x model log likelihood of parameters (q) given the data + 2 Nro. Parameters. The model with the lowest AIC value was better supported by the data and more parsimonious than other models. Since the dataset is small, MARK calculated the corrected AIC or AIC_c. Since the candidate model set was adjusted by using the c-hat obtained from the GOF test, MARK displayed the quasi AICc or QAICc values.

Delta AIC (ΔAIC): the difference between each model and the one with the lowest AIC value (the most parsimonious). MARK calculated ΔAIC_c because of the small sample size, and $\Delta QAIC_c$ after adjustment. The following rules of thumb were applied to determine what models were different or not: (a) If $\Delta QAIC_c < 2$, both models have equally weight in the data. No real difference between the two models, (b) if $2 < \Delta QAIC_c < 7$, there is considerable support for a real difference between the two models, and (c) if $\Delta QAIC_c > 7$, there is a strong evidence to support to the conclusion of difference between the two models.

The rule of thumb above was used in combination with the normalized Akaike weights. The weight (***wi***) for each model was calculated with Equation 4-1:

$$wi = \frac{\exp\left(\frac{-\Delta Q\mathrm{AIC_c}}{2}\right)}{\sum\left\{\exp\left(\frac{-\Delta Q\mathrm{AIC_c}}{2}\right)\right\}} \tag{4-1}$$

So, wi is the proportion of the data that support a particular model in comparison with all models. The model with the highest ***wi*** would be the best model because it had more support than any other model. But, to know how much better it was than the next model, the ***wi*** of the best model was divided by the ***wi*** of the next best model. This quotient stated how much the best model was supported by the data than the next best model.

Model likelihood (index of relative plausibility): the ACI weight of the model of interest divided by the ACI weight of the best model. This quotient indicated how likely a particular model was in comparison with the best model. It is important to highlight that the AIC approach helps to select the best model; however, there is an uncertainty of which model is the best model. The normalized Akaike weights and the likelihood of the model measure this uncertainty.

The number of parameters and model deviance were the last two criteria. The model deviance is the difference in $-2\ln \pounds\left(\mathrm{q/data}\right)$ of the current model and $-2\ln \pounds\left(\mathrm{q/data}\right)$ of the saturated model. The saturated model is the model with the number of parameters equal to the sample size.

Second, *P. floridanus*, the most parsimonious model was phi(t) p(.), and the second and third best models were phi(t) p(g) and phi(t) p(t), respectively. In *S. hispidus*, the most parsimonious model was phi(t) p(.) and the second one was phi(t) p(g) The survival probability (phi**)** was only time dependent, and the probability of recapture (p**)** was constant (**.**), time dependent (t), and group dependent (g). Therefore, I considered building other models to test the prescribed burning and other effects based on these preliminary results.

Third, adding models phi(Flood + Fire) p(., t, g) for *P. floridanus* and phi (Flood + Fire) p(., g) for *S. hispidus*. Flood and prescribed burning (Fire) are time dependent variables and their additive effects were modeled with the probability of recapture constant (**.**), time dependent (t), and group dependent (g). This combination resulted in nine models for *P. floridanus* and six models for *S. hispidus* that were added to the candidate model set of 16 models for comparison purposes. Then, the most parsimonious model was selected out of 25 and 22 models for *P. floridanus* and *S. hispidus*, respectively. The covariate Flood was included because three hurricanes hit Cedar Key and partially flooded treatment and control sites. No previously marked mice were recaptured after flooding. Since, this covariate had a strong influence in the survival probability in both species; its additive effect with prescribed burning was modeled by adding linear constraints to MARK. The basic sequence of steps of building design matrices followed Cooch and White (2006).

Fourth, checking for the number or real parameters and adjusting them. Even though MARK estimates the number of parameters, the model structure determines the number of parameters that are theoretically estimable. However, if the sample size is small, not all theoretically estimable parameter can be estimated. In addition, when survival and recapture are time dependent, the terminal parameter is not individually identifiable (Lebreton et al. 1992). Since the sample size for the mark-recapture data is small in this study, I manually checked the number of estimable parameters indicated by MARK matched the number of β parameters theoretically estimable for a particular model. If they did not match (one or more β parameters were not estimated), I manually adjusted the number of parameters to the theoretical number.

Fifth, the model selection carried out in the previous steps allowed the testing of the flood and prescribed burning effect on the survival probabilities in both species and to estimate survival and recapture parameters. The most parsimonious model provided the most precise and less biased survival estimates. However, since there was an uncertainty about which model was the best model; there was also an uncertainty with survival values. Reporting survival estimates from a single model in the candidate model set, even if it was the most parsimonious model, ignored model uncertainty. For this reason, survival estimates were reported by using modeling averaging, which allowed estimating the average of each parameter of interest from the model set. Modeling averaging takes the estimates of various models, and weights them by using the normalized AIC weights. The following equation was used (Equation 4-2):

$$\text{avg}\,(\hat{\phi}) = \sum_{i=1}^{R} \text{wi}\,\hat{\phi}i \tag{4-2}$$

The average value for the parameter $\hat{\phi}$ was calculated by multiplying the AIC weight of model i and the parameter value estimated by that model and adding up all these products in the model set. Finally, survival estimates from both the most parsimonious model and model averaging were plotted. The purpose of this comparison was to show similarities or dissimilarities between estimated parameter from both approaches.

Results

Number of Captured Individuals in Treatment and Control Sites

A total of 182 individuals were marked and recaptured 426 times in 29,340 trapping nights. Figure 4-2 illustrates the number of captured individuals per species per trapping session in treatment sites. All species were captured with low numbers (1-9 individuals) at the beginning of the study. Of 39 individuals marked in the first 2 trapping sessions, I did not recapture any of them after the hurricanes during the 3rd trapping session and during the additional trapping effort carried out in upper grounds. Most likely, they died. After the 3rd trapping session, the number of individuals of *S. hispidus* and *P. floridanus* increased through time from 13 to 34 individuals and from 13 to 21 individuals, respectively, even after prescribed burning. Also, after the 3rd trapping session, the number of captured individuals of *P. gossypinus* was stable (4-7 individuals) even after prescribed burning. The number of captured individuals of *O. nuttalli* did not increase after prescribed fire, and only one individual was recaptured through time.

Figure 4-3 shows the number of captured individuals per species per trapping session in control sites. Again, all species were captured in low numbers (1-11 individuals) at the beginning of the study. Of 39 individuals marked during the first 2 trapping sessions, I did not recapture any of them after the hurricanes during the 3rd trapping session and during the trapping done in upper grounds. Probably, they also died. I captured 9 and one new individuals of *P. floridanus* and *S. hispidus*, respectively, during the 3rd trapping session. After the 3rd trapping session, the number of individuals of *S. hispidus* increased from 2 to 13 individuals, and the number of captured individuals of *P. floridanus* (6-8 individuals) and *P. gossypinus* (3-6 individuals) remained relatively stable through time. Only one individual of *O. nuttalli* was recaptured after the 3rd trapping session.

Figures 4-2 and 4-3 present two different patterns. The number of individuals of *S. hispidus* and *P. floridanus* increased in treatment sites only, and this event took place after flooding and prescribed burning. In contrast, a similar pattern was found for *P. gossypinus* and *O. nuttalli* in treatment and control sites. Obviously, flooding, prescribe burning, or both affected demography parameters of *P. floridanus* and *S. hispidus*.

The extra trapping effort carried on for control sites in June-July 2005 indicated that mice did not move because of mowing. Mowing the vegetation took 2-3 weeks in each site, and the mechanical activity was very intense. Even though the mechanical activity produced a loud noise that could be heard from 500 m, marked mice were not affected by this type of perturbation because they remained in the grids.

Number of Captured Individuals in Scrubs and Wetlands

Table 4-1 presents the number of captured individuals in the scrubby flatwoods (scrubs from now on) and in the vegetation surrounding wetlands (wetlands from now on) per trapping session in treatment and control sites. In treatment sites before prescribed burning (1^{st} - 4^{th} trapping sessions), 54 (74%) out of 73 individuals were captured in scrubs (see also Figure 4-4). During the 4^{th} trapping session after the hurricanes, 26 (77%) out of 34 new marked individuals were captured in scrubs, and 22 (85%) out of the 26 mice were recaptured in wetlands during the 5^{th} trapping session right after prescribed burning. During the 5^{th} trapping session in wetlands, five mice previously trapped in wetlands were recaptured and 18 new individuals were captured. Therefore, marked mice moved to or stayed in wetlands after prescribed burning and new individuals preferred wetlands rather than scrubs. During the 6th and 7th trapping sessions, 90 individuals were captured in wetlands and six in scrubs. The 90 individuals included mainly previously marked individuals (81) rather than new ones (9). The six individuals found in scrubs corresponded to one *P. floridanus* marked in the 6^{th} and recaptured in the 7^{th} trapping session, two new *P. floridanus* marked in the 7th session, and one *P. gossypinus* recaptured in the 6th and 7th trapping sessions. Of 46 mice captured in wetlands during the 7th trapping session, 31(67%) were recaptured in scrubs during the 8th trapping session. During this last trapping session, one year after prescribed burning, 53 (90%) out of 59 captured individuals were trapped in scrubs. The 53 individuals included 31 recaptured mice from the 7th trapping session and 22 new individuals. Also, six new individuals were captured in wetlands. During all trapping sessions in control sites, 153 individuals were captured, from which 139 (91%) mice were captured/recaptured only in scrubs (Table 4-1 and Figure 4-5). Therefore, mice returned to the scrubs in treatment sites after at least 11 months (May 2005-April 2006) following prescribed burning. These results clearly supported the research hypothesis and indicated that the vegetation surrounding wetlands provided refuge to the small mammals for at least 11 months following prescribed burning.

Mice returned to scrubs after plant species offered both cover and food, and this event took place at least 11 months after prescribed burning. The 7th trapping session occurred in November 2005, and the majority of the mice were still in wetlands. I could not trap at nine months after burning (January 2006) because of the low temperature. Trapping started in April 2006 at 12 months after burning and mice were recaptured in scrubs. Therefore, mice may have moved to scrubs in March 2006 or after it. Mice probably did not move to scrubs in January-February because insect activity was assumed to be low due to the low temperatures and plant species did not start to produce flowers and fruits until April-May 2006. Most likely, mice found both cover and food in scrubs after 11 months following prescribed burning and returned to scrubs for this reason.

Flood and Fire Effects on *Podomys floridanus*

The GOF test for *P. floridanus* is presented in Table 4-2. The Bootstrap c-hat method did not give any results, which may be due to the small sample size, and the Bootstrap deviance method provided a c-hat larger than the c-hat obtained from the Release method. Hence, the general model and the candidate model set were adjusted to the c-hat = 1.1387 in order to be conservative. This c-hat indicated that there was a little of overdispersion in the dataset.

The additive effect of Flood and Fire had an effect on the survival probability of *P. floridanus*. Table 4-3 presents the candidate model set of 16 models adjusted to c-hat = 1.1387. Model phi(t) p(.) was the most parsimonious model with 73.16% support in the data. But, because of the first three models had 99.88% support in the data, phi in these models was time dependent, and p was constant (.), group dependent (g), and time dependent (t), I modeled phi with the covariates Flood and Fire in combination with p (., g, t). Table 4-4 displays the 25 models after including nine models from the combination phi(Flood + Fire) p(., g, t) and correcting for the number of parameters. As shown in this table, the top model phi (Flood + Fire) p(.) has only 29.92% support in the data, and there is no enough evidence to indicate that this model is different from the 2nd to the 5th model because delta $QAIC_c < 2.0$. However, the additive effect of Flood + Fire and the covariate Flood had a strong influence on the survival probability of *P. floridanus* because this set of models is supported by 61.67% of the data, and phi is dependent of Flood + Fire and Flood in the first four models. Even though the covariate Fire by itself does not have support in the data, the effect of fire can be seen by comparing model phi (Flood + Fire) p(.) with phi (Flood) p(.). Delta $QAIC_c = 2.83$, and this is the Fire effect. The effect of Flood can be noted by comparing phi (Flood + Fire) p(.) with phi (Fire) p(.). Delta $QAIC_c = 30.11$. This is the effect of Flood. Model phi(Flood + Fire) p(.) estimated one survival and one recapture parameters because this was a particular case of time-dependence where trapping sessions with the same Flood and Fire conditions shared the same survival rate (Table 4-5). MARK does not count parameters with standard errors equal to zero in the parameter total. So, the survival rate for non-flood and non- prescribed fire times was 0.7871 in treatment and control sites, and 0.00 and 1.00 for flooding and after prescribed fire times, respectively. Therefore, the additive effect of Flood + Fire had an influence on the survival probability of *P. floridanus*.

The time-dependent covariate Flood and Fire did not have an important influence in the recapture probability. Because models phi(Flood + Fire) p(t) and phi(Flood) p(t) had 31.76% support in the data and p(t) was present in these two models, I added models phi(Flood + Fire) p(Flood+Fire), phi(Flood + Fire) p(Flood), and phi(Flood + Fire) p(Fire) to analyze the effect of the time-dependent covariates Flood and Fire in the recapture probability. Table 4-6 presents these set of models, and each of them had very little support in the data (< 8.7 %). Consequently, model phi(Flood + Fire) p(.) still was the most parsimonious model.

Prescribed burning did not have a negative effect and flooding probably negatively influenced the survival probability of *P. floridanus*. The results of the current analysis indicated that phi(Flood + Fire) p(.) was the most parsimonious model among 28 models. The survival parameters estimated for treatment and control sites from this model revealed that both curves were similar, and the only difference was for phi_4, where prescribed burning apparently increased survival (Figure 4-6). But, I cannot conclude if this increase was significant or not because the β parameter was not estimable (Table 4-7). Nevertheless, I can state that prescribed fire did not have a negative influence on the survival probability of *P. floridanus* because any of the models in which Fire was involved alone had support in the data. Furthermore, in the real scenario, prescribed burning indirectly increased survival because of the role of the vegetation surrounding wetlands as refugia. Thus, these results support the research hypotheses. In contrast, flooding probably had a negative effect. The β parameter in the model phi(Fire + Flood) p(.) was not estimable. Statistically, I cannot make any conclusion. But, practically, it is likely that Flood decreased the survival probability of *P. floridanus* to zero before the 3rd trapping session. I did not recapture 31 *P. floridanus* marked between the 1st and 2nd trapping sessions after the hurricanes.

Table 4-8 summarizes the estimated survival parameters for model averaging, and Figure 4-7 displays the estimated survival parameters for model phi(Flood + Fire) p(.) and model averaging in treatment and control sites. The fact that four curves were quite similar indicated that model phi(Flood + Fires) p(.) was the best model and provided pretty good estimates of the survival parameters. The main difference was on phi_4, in which the two estimates in treatment sites differed by 0.032212. However, model averaging provided a better estimate on phi_4 because of 13 *P. floridanus* marked in the 4th trapping session, 12 were recaptured in the 5th trapping session. So, phi_4 should not be equal to1.00.

Flood and Fire Effects on *Sigmodon hispidus*

Table 4-2 shows the GOF test for *S. hispidus*. The Bootstrap deviance method gave a c-hat larger than the c-hat obtained from the Release method, and the Bootstrap c-hat method did not provide any result probably because of the small sample size. The general model and the candidate model set were adjusted to the c-hat given by the Bootstrap deviance method (c-hat = 1.5694). This c-hat revealed overdispersion in the dataset.

The covariate Flood and the additive effect of Flood and Fire had an influence on the survival probability of *S. hispidus*. The candidate model set of 16 models adjusted to a c-hat = 1.5694 is shown in Table 4-9. The most parsimonious model was phi(t) p(.) with 69.78% support in the data, and the first two models had 95.59% support in the data. The survival probability phi in these models was time dependent, and the recapture probability p was constant (.) and group dependent (g). Consequently, I decided to model phi with the time-dependent variables Flood and Fire in combination with p (., g). The combination of these models produced six models that were added to the candidate model set. Table 4-10 shows the 22 models fitted, adjusted to c-hat = 1.5694, and corrected for the number of parameters. As can be seen in this table, the top model, phi (Flood) p(.) had 32.51% support in the data, but it was no different from the 2nd to the 4th model because delta $QAIC_c < 2.0$. Of the remaining 18 models, phi(Flood + Fire) p(g) and phi(t) p(g) had 9.96% and 5.15% support in the data, respectively, but the rest of the models had a support ≤ 0.37% in the data. In addition, there were considerable evidences for a real difference between phi(Flood) p(.) and the 5th and 6th models because $\Delta QAICc > 2.0$. The first four models had 83.75% supports in the data and it is conformed mainly by the covariates Flood and Flood + Fire. Thus, only models with Flood and Flood + Fire on the apparent survival rate had a substantial support in the data. Therefore, these covariates had an effect of the survival probability of *S. hispidus*.

The covariate Flood and the additive effect of Flood and Fire apparently reduced the survival probability of *S. hispidus*, but Fire did not have any negative effect on the survival rate. The most parsimonious model phi(Flood) p(.) and the second most parsimonious one phi(Flood + Fire) p(.) had 32.51% and 23.77% support in the data, respectively (Table 4-10). In both models, Flood apparently decreased survival from values such as 0.9082 and 0.9276 to 0.0000 between the 2nd and 3rd trapping sessions (see Table 4-11, Figure 4-8). Also, Fire in model phi(Flood + Fire) p(.) apparently decreased survival from 0.9276 to 0.7736 between the 4th and the 5th trapping sessions. The word 'apparently' is used because the β parameters corresponding to Flood in the two most parsimonious models were not estimable, and Fire in model phi(Flood +

Fire) p(.) was estimable but not significant (Table 4-12). Practically, Flood probably killed all marked *S. hispidus* in the study during hurricanes even though it could not be demonstrated statistically. Six *S. hispidus* marked during the first two trapping sessions were not recaptured after the hurricanes. In contrast, the covariate Fire had ≤ 0.186% support in the data in the three models that stood alone, and it was not significant (Table 4-12). Therefore, prescribed burning did not have a significant negative effect on the survival probability of *S. hispidus*. Out of 13 marked rats during the 4^{th} trapping session, 10 were recaptured in the vegetation surrounding wetlands after prescribed burning. Hence, these results support the research hypothesis.

The estimated survival parameters for model phi(Flood + Fire) p(.) and model averaging are presented in Table 4-11 and Table 4-13, respectively, and Figure 4-9 displays the comparison. Even though phi(Flood + Fire) p(.) was not the most parsimonious model, it was compared with model averaging because this model and model phi(Flood) p(.) had equal weight in the data, and phi(Flood) p(.) did not contain Fire and model averaging did. The survival parameters estimated by model phi(Flood + Fire) p(.) were the same for treatment and control sites, with the exception of phi_4, because this was a case of time-dependence where trapping sessions with the same flood conditions would share the same survival rate. These estimated parameters were similar to the parameters estimated by modeling averaging with the exception of phi_3 and phi_4. For $phi_{3,}$ the value estimated by model phi (Flood + Fire) p(.) in treatment and control sites (0.9275804) was higher than the value estimated by modeling averaging in treatment (0.8513742) and control (0.8512919) sites. Nonetheless, these last two values were so similar that are shown as one point in Figure 4-9. For phi_4, model phi(Flood + Fire) p(.) estimated a value for treatment (0.7736407) lower than for control sites (0.9275804). Model averaging also produced a value for treatment (0.8413499) lower than for control (0.9037845) sites. But, the difference between treatment and control sites for model phi(Flood + Fire) p(.) was higher (0.1539397) than the difference for model averaging (0.0624346). Therefore, model phi(Flood + Fire) p(.) provided good estimates of the survival parameters with the exception of phi_3 and phi_4. Parameter $phi_2 = 0.00$ because six rats marked during the first two trapping sessions were not recaptured after the hurricanes during the 3rd trapping session.

Trapping Predators

An increased in the number of predators was noticed in all sites before the 8th trapping session, but mainly in treatment sites. Before prescribed burning, *P. lotor*, *U. cinereoargenteus*, *D. marsupialis*, and *M. frenata* sprung traps for no more than three days. The first three species were identified by their tracks, and the Florida long-tailed weasel was seen on 04/06/05 at 5:37 pm hunting in the grid in 5A. These predators moved to other places after finding traps closed for 3-5 days. However, before the 8th trapping session and coincidently with the return of mice to the scrubs, predators were trapped in all sites (Table 4-14), and they did not move to other places after closing traps for 3 days. This problem was solved by trapping predators before small mammals for 7-10 days and translocating them to other places at least 3.0 km away from the grids.

Discussion

Low Capture Success

Morgan's study (1998) and the current study shared sites. Morgan trapped on transects and at burrows in three places named CKA, CKB, and CKC. CKA and CKC are in the 5A site in this study; but, transects installed in CKA and CKC and the grid installed by this study were in different places. CKB is the same 5C site in the present study, and trapping occurred at the same place. Even though trapping did not occur in the exactly same place in 5A site, trapping took place in scrubby flatwoods in the same study area. Sites 5A and 5C were not burned since 1957

(Morgan 1998), and 2M and 5D did not have records of the last burn, but apparently both sites burned in a wildfire in 1955 (Jeff DiMaggio, per. com.). Morgan trapped in 1997-98, at least 7 years prior to my trapping period and she obtained a high trapping success.

This study found a low capture success even though a major trapping effort was carried out. Table 4-15 shows the trapping effort of several studies conducted for small mammals in Florida. Jones (1990) had the highest with 33,000 trapnights. The current study had the next highest with 29,340 trapnights. Nonetheless, Jones trapped during 60 months, and I trapped during 31 months. Morgan trapped 2,970 trapnights, and she reported a higher number of individuals than the present study. Morgan trapped 430 *P. floridanus*, 275 *S. hispidus*s, 221 *S. hispidus*, and 33 *O. nuttalli*. I expected to obtain a high capture success due to Morgan's capture success in the same study area. In addition, Layne (1990) reported peaks in populations in late winter and early spring, exactly when I did the first trapping session. There was a lapse of at least 6 years between Morgan's and the current study. During this time, vegetation density increased and structure changed with all the consequences for small mammals' habits and behaviors associated with these changes.

The age of the vegetation is an important factor because all sites were not burned for at least 47 years. When fire suppression leads to habitat conversion, *P. floridanus* populations tend to be reduced or eliminated (Layne 1990, 1992, Jones 1992). Populations of *P. floridanus* in scrub habitat have shown to decline with increased vegetation density (Layne 1990, 1992). However, this explanation is not valid for *S. hispidus* and *P. gossypinus* since they are opportunistic and able to use other types of habitats (Layne 1974, Eisenberg 1983). But, vegetation succession without fire with the corresponding increase in vegetation density might explain a high capture of *O. nuttalli* in comparison with Morgan's study. The vegetation in all sites may be more suitable for *O. nuttalli* due to its arboreal habit.

Fire suppression also decreases habitat suitability for *Gopherus polyphemus* (Cox et al. 1987), and this aspect influences *P. floridanus* presence in all sites. Even though *P. floridanus* has been cited as a facultative user of gopher tortoise's burrows in scrubby flatwoods (Morgan 1998), the high capture success found in burrows by Morgan was an indication of the close association between *P. floridanus* and the gopher tortoise's burrows in this type of habitat. Of 92 burrows found in four sites in this study, only 13 were gopher tortoise's burrows. It is very likely that 79 burrows were *Dasypus novemcinctus* (armadillo) burrows because of the circular shape of the entrances, and traps placed near entrances were not removed.

Small mammals have natural population fluctuations through time. Particularly, population cycles have been reported for small mammals (Batzli 1992). It is possible that trapping occurred when the population size for these species was low. A low mast production could be responsible for a low population level, but there were no data to support this statement. However, I observed that acorn production was low during 2004-2006 in all sites.

Vegetation age, fire suppression, population cycles, and acorn production or a combination of these factors probably influence capture success. A combination of these factors maybe worked together. Other factors such as trapping design and type of bait were not the causes of the low capture success. The 10x10 grid with traps every 15 m has been used as the standard grid design in the majority of the small mammal studies in the United States (John Eisenberg, per. com.). In general, scientists with experience trapping small mammals in Florida consider trapping success low on grids and between 1% and 5% (John Eisenberg and James Layne, per. com). However, if we compare Morgan's trapping design (three U-shaped loops of 210 m long with 14 trapping station separated by 15 m and 45 m between loops) with the current study, loops cover a smaller area (1.42 ha) than grids (2.25 ha), so I would expect to have a higher capture success on grids, and Morgan's results were evidence of the opposite. Morgan used crimped oat as bait with an excellent result. I used a 50-50 mixture of crimped oat and sun flower seeds, which probably is much better bait than crimped oat by itself. I found that *P. floridanus*, *S. hispidus*, *P. gossypinus*, and *O. nuttalli* ate all the sunflower seeds and not all the crimped oats in the traps.

Flooding Effect

Hurricanes Charley (9-14 August), Frances (25 August – 8 September), and Jeanne (13-28 September) hit the Florida peninsula in 2004. Charley did not hit Cedar Key directly, but its winds brought rainfall to the area. Frances and Jeanne hit Cedar Key directly and brought a high precipitation into the area. A total of 372.5 mm fell in Cedar Key during September (data from http://www.AccuWeather.com). This is 217.6 mm over the monthly average precipitation (154.9 mm). This amount of precipitation inundated wetlands, scrubby flatwoods, and sand pine scrub. All sites were partially flooded. The approximate percentage of the grids covered by water was as follows: 5C = 50%, 2M = 60%, 5A = 50%, and 5D = 30%. All sites stayed flooded for at least 2 weeks. Could small mammals survive this amount of precipitation? Did they move to upper grounds?

Mice only had two options, move to upper ground or die. All sites had upper grounds inside the grids; therefore, the 3rd trapping session already sampled upper grounds in all of them. The extra effort involved the upper ground in/near the grids. However, no mice were captured during this time. The soil in scrubby flatwoods and sand pine scrubs could not absorb the amount of precipitation that fell in Cedar Key during the impact of the hurricanes. Thus, a high portion of the preserve was flooded, and this condition strongly impacted the small mammal community.

Most likely, 79 terrestrial and arboreal mice marked during the first two trapping sessions in treatment and control sites died. I do not think that mice had time to move to other upper ground areas not covered by traps.

Flooding negatively affected the small mammal community independently of their life form. Even though the effect of flooding on the survival probabilities of *P. floridanus* and *S. hispidus* was not conclusive, the effect of flooding upon terrestrial species such as *P. floridanus*, *S. hispidus*, and *P. gossypinus* was expected to be detrimental to their populations. Arboreal species such as *O. nuttalli* should have a better possibility to tolerate this type of disturbance, but it did not because flooding lasted at least 2 weeks in all sites. No previous survival analysis of the effect of flooding on small mammals was found, but some references support the negative effect of a long period of flooding. No detrimental effect upon the population of *P. gossypinus* and *O. nuttalli* in Texas was recorded when flooding occurred up to 8 days. Flooding for a 3-week period caused a marked decrease in the populations. This probably happened because individuals tended to remain within established home range even during long periods of flooding (McCarley 1959). *Peromyscus leucopus* (white-footed mice) completely disappeared from floodplain plots after severe flooding (Blair 1939, Turner 1966). *P. leucopus*, *Microtus montanus* (mountain vole), and *Dipodomys ordii* (kangaroo rat) generally experience habitat inundation as catastrophic (Andersen et al. 2000).

Owls hunted snakes during day light hours after the impact of the hurricanes. I observed three owls hunting near wetlands between 3 pm and 4 pm in 5C and 5D. Probably, the species was *Bubo virginianus* (great horned owls) because of the big size and ear tufts. Two of them already caught a snake. This species is mainly nocturnal and eats rodents. Even though these observations are not enough to drive a conclusive statement, maybe owls were hunting snakes during light hours because of the reduction of small mammal populations due to the impact of the hurricanes.

Population Responses during and after Prescribed Burning

Small mammals hide in burrows to survive and respond to change in cover and food caused by prescribed burning. Each species, independent of the life form, must retreat to burrows to survive the passage of the fire front, and it is likely to respond differently to prescribe burning and subsequent habitat changes. Some species increase in population and others decrease in populations after prescribed burning. Other species just disappear from burned areas. Some species avoid recent burns until habitat requirements removed by prescribed burning are restored. These aspects are discussed in more detail in the following subtitles.

Burrows as refugia during prescribed burning

Probably, the stand-replacing prescribed fire did not cause mortality of the small mammal community in CKSSR because mice hid in burrows. Fire intensity should be equal in treatment sites from the experimental point of view. However, this aspect was not as important as expected because fire intensity was high enough in both sites to remove all above ground vegetation. No evidence of mice mortality caused by prescribed burning was found after a careful search of both sites, and this result might be a consequence of the behavior of the species during prescribed burning. *P. floridanus* and *P. gossypinus* are nocturnal species that live in burrows and have to stay in them to survive the passage of fire. *S. hispidus* is a diurnal-nocturnal species, and if individuals are active during the day, they have to move and to hide in burrows or to flee if their home range is burned. *O. nuttalli* is arboreal and nocturnal species that live in above ground nests and would have to seek refuge in burrows to survive. There were a total of 28 burrows in each treatment site. Out of 33 individuals of *P. floridanus*, *S. hispidus*, and *P. gossypinus* marked during the 4th trapping session, 26 individuals were recaptured in wetlands after prescribed burning during the 5th trapping session. Only one individual of *O. nuttalli* was marked during the 4th trapping session, and it was recaptured in wetlands after prescribed burning in the 5th trapping session. Probably this individual also hid in a burrow because otherwise it would not survive. Burrows increase survivorship because of the insulating characteristic of the soil. Some studies have shown that temperatures higher than 100 °C at the surface of the ground decline in the first 2.5 cm of soil depth to temperatures around 20-30 °C in longleaf pine in south-eastern USA (Heyward 1938), in Australia eucalypt forest (Beadle 1940), in California Chaparral (DeBano et al. 1998), and in heavy slash fuels after logging a forest (Neal et al. 1965; cited by Whelan 1995). One reason for poor penetration of heat is that convective heat is transferred upward.

Burrows as refugia for small mammals during prescribed fire have been documented in the literature. Small mammals can survive fires by remaining in their burrows (Tester 1965, Beck and Vogl 1972; Hendlund and Rickard 1981; Smith 2000). Most species look for refugia underground, where ventilation inside burrows is vital for animal survival (Bendell 1974). In this regard, burrows with more than one entrance might be better ventilated than those with one entrance (Geluso et al. 1986). No mortality of *P. floridanus* was found in two sites in Ordway-Swisher Preserve possibly because of the uneven distribution of litter and bare patches of sands caused a mosaic effect of varying intensities, and mice were protected in the tortoise burrows (Jones 1990). However, Jones also reported 87% return rate in one site, and probably, some mortality occurred after prescribed fire in this site. Burrowing rodents, such as *Dipodomys*, survived in substantial numbers after a stand-replacing fire in California chaparral because their burrows protected them from heat (Quinn 1979). Populations of Townsend's ground squirrels living in burrows were unaffected by stand-replacing fire in sagebrush-grass community in southeastern Washington (Hendlund and Rickard 1981). Regarding arboreal species, woodrats usually suffered relatively high mortality because their nests were above ground (Simons 1991). However, populations of woodrats were "unexpectedly high " in burned areas because burns left patches of "lightly burned "vegetation in California chaparral and coastal sage scrub, which may have provided refugia for woodrat populations (Schwilk and Keeley 1998). In summary, during fire, the majority of the species seek refugia underground or in sheltered places above the ground.

Prescribed burning effect

The lack of a negative prescribed burning effect found in this study has also been cited in the literature. Even though three studies have reported survival analysis by using information-theoretic model selection and inference framework through the program MARK, two of them indicated that prescribed burning did not have any effect on the species involved. On all plots, independent of shrub density or burn treatment, the abundance of *Neotoma fuscipes* (dusky-footed woodrats) increased from 1993 to a peak in 1997, and decreased from fall 1997 to fall 2001 after prescribed fire in California oak woodlands. Apparently, juvenile survival appeared to be the cause of the population fluctuation in this species. Prescribe fire by itself did not have any support in the data (Lee and Tietje 2005). In a single prescribed fire in old growth mixed conifer forest in Sequoia National Park, California, where fire had been suppressed for over a century, year effects had greater influences on *P. maniculatus* densities, *P. maniculatus* age ratios, *Neotomias speciosus* (lodgepole chipmunk) densities, and total small mammal biomass than did prescribed fire effects. Fire by itself had less than 0.01% support in the data (Monroe and Converse 2006). In ponderosa pine in Coconino National Forest, Arizona, forest thinning increased densities of *P. maniculatus*, *Tamias cinereicollis* (gray-collared chipmunks), *Spermophilus lateralis* (golden-mantled ground squirrels), and *Neotoma mexicana* (Mexican woodrats), but the combination of thinning and frequent prescribed fire might have reduced small mammal densities (Converse et al. 2006). Prescribed burning was not applied as a treatment by itself.

Population increases/decreases after prescribed burning

One important aspect to take into consideration for the experimental design is the homogeneity of treatment and control sites regarding soils, slopes, plant and small mammal species composition, vegetation structure, and fire history. Particularly, the normal seasonal variation in numbers of individuals before prescribed burning should be similar in treatment and control sites in order to detect prescribed burning effects. A high proportion of the studies have this problem. The current study attempted to determine this aspect by trapping four times before and after prescribed burning. The interval of time before and after was 13 and 15 months, respectively. However, the flooding effect caused by the hurricanes did not allow determining the normal seasonal variation in experimental sites before burning. This natural disturbance was recorded in conjunction with prescribed burning, which makes the current study unique in this sense. The effect of flooding previously presented was different from the effect of prescribed burning.

There was an increase in the number of individuals of *P. floridanus* and *S. hispidus* after prescribed burning. The increase in the number of individuals of *P. floridanus* and *S. hispidus* was clearly identified when I compared treatment and control sites (Figures 4-2 and 4-3). There was no doubt that prescribed burning forced these individuals to move to the vegetation surrounding wetlands. However, *P. gossypinus* and the only individual of *O. nuttalli* did the same, but the number of individuals did not increase through time. So, prescribed burning was the stimulus to move, but the increase in number of individuals in the vegetation surrounding wetlands had to deal with other factors such as immigration, food/space availability, and competition. This dissertation only has data for the first factor. The increase in the number of individuals was due to new adult individuals. No juvenile was captured between the 5^{th} and the 7^{th} trapping session. So, apparently, the burned area attracted these mice but they had to seek refuge in the vegetation surrounding wetlands because of the lack of cover and food in the burned area.

The lack of increase in the number of individuals of *P. gossypinus* was surprising, but the drastic decline in the number of individuals of *O. nuttalli* was expected. The majority of the studies have shown a positive response of the genus *Peromyscus* to prescribed burning (Jones 1992). It was astonishing to see that the number of individuals of *P. gossypinus* that maintained relatively stable low population levels in treatment and control sites (Figures 4-2 and 4-3). Maybe, competition for food and space did not allow new *S. hispidus* to establish in the vegetation surrounding wetlands. However, the decline in the number of individuals of *O. nuttalli* was predictable because of its arboreal life form and the combined effect of flooding and prescribed burning. Both curves in Figures 4-2 and 4-3 look alike. *O. nuttalli* appears to be susceptible to prescribed burning even though it might hide in burrows to survive the path of fire.

The increase, decrease, or no change in the number of individuals after prescribed burning has been reported by other studies conducted in Florida. *P. floridanus* and *P. polionotus* increased population after prescribed fire while *S. hispidus* declined. However, no change in the composition of the populations of *P. floridanus* and *S. hispidus* was recorded before and after burning in longleaf/turkey oak habitat in north-central Florida (Arata 1959). Densities of *P. gossypinus* and *S. hispidus* appeared to be similar in the burned and unburned hardwood forest in north Florida (Vogl 1973). Three populations of *P. floridanus* had little or no mortality due to prescribed fire, and populations were higher on burned areas than on unburned sites in longleaf/turkey oak habitat in Ordway-Swisher Preserve. This means that *P. floridanus* did not move from the burned area after prescribed burning. Numbers of mice were more constant in the burn site than in the unburned one. Significantly more mice were caught in burned areas immediately following prescribed fire. Also, significantly more mice were caught on burned sites than on unburned burrows (Jones 1989, 1990). *P. floridanus* populations appeared to have little or no short term effects following prescribed burning, and abundance equaled or exceeded pre-fire levels after two or three months (Jones 1992). *P. gossypinus* and *S. hispidus* had an increase in the number of individuals after a wildfire in slash/longleaf pine habitat in north-central Florida (Layne 1974). Layne also suggested that burned areas could act as "dispersal sinks." This statement might be true in some ecosystems, but not in others. In the current study in CKSSR, four *P. floridanus* and two *P. gossypinus* were recaptured only in the scrubs after prescribed burning. Layne (1990) also carried out a long-term monitoring of *P. floridanus* population in CKSSR. He reported that the species survived a wildfire in 1955, population declined 10 years after fire, and the species was still present in 1986. Another study conducted by Layne (1990) at Archbold Biological Station revealed that *P. floridanus* was present at low numbers in scrub sites that were burned in 1927. In contrast, populations were higher and more stable in similar nearby habitats that were burned periodically. According to Layne (1992), *P. floridanus* populations are higher in early successional stages of scrub vegetation following fire. Particularly, high population numbers can be found in 2 years old scrub (Layne, personal communication). Later in the absence of fire, populations decline as habitat structure becomes denser and microclimatic conditions more mesic. However, Morgan (1998) reported a high capture success between 1997 and 1998 for *P. floridanus*, *S. hispidus*, *P. gossypinus*, and *O. nuttalli* in CKSSR, and the current study found a low capture success between 2004 and 2006. This change in population size can be explained by natural population fluctuations, and probably the high population abundance found by Morgan after 42 years might be related with mast production. Depue (2005), in central Florida, found that *P. floridanus* increased or recovered pre-burn levels within 6 months following prescribed burning in Bullfrog Creek Mitigation Park; it dropped in numbers following prescribed fire, but started to increase when the study ended in Split Oak Mitigation Park; and the decrease in animal numbers remained unaffected by prescribed fire in Chuluota Wilderness Area. Apparently, there is not a cut-clear pattern by a single species in Florida, and some variation in the response to prescribed burning might occur within the same region. Therefore, the same variation might be expected in other studies conducted in other states.

Some species increase population and others decrease populations after prescribed fire in other studies conducted in the United States. *P. maniculatus* had a post-fire increase in population size in California chaparral (Cook 1959) and in northeastern Minnesota (Ahlgren 1966), and these incrases were probably related to the increase of seeds of annual grasses stimulated by fire. Prescribed burning reduced the population of small mammal species with the exception of *P. maniculatus* in north-central Pennsylvania. The *P. maniculatus* established in the burned area one month following prescribed burning (Fala 1975). Populations of *P. maniculatus* generally increase after fire (Ream 1981). *P. maniculatus* were more abundant on 1-2-year-old burns in tallgrass prairie than in unburned areas in eastern Kansas. However, *Reithrodontomys megalotis* (western harvest mouse) was more abundant in unburned areas (Kaufman et al. 1982). Also, Kaufman et al. (1983) reported that *R. megalotis* densities in the burn site increased the following spring and summer because the population in the un-burned sites served as a source of dispersing individuals. *P. maniculatus* dramatically increased population size one year following prescribed burning; however, this effect disappeared and reversed itself during the second year in Ponderosa Pine, South Dakota (Bock and Bock 1983). The total number of small mammals was lower in burned sites than in unburned ones in shrub-steppe habitat in Idaho. Nonetheless, *P. maniculatus* were more abundant in the burn site (Groves and Steenhof 1988). A dramatic increase in *P. maniculatus* population on burn sites following prescribed fire in British Columbia was explained by the rodents' ability to forage for seeds and insects that were greatly increased. In contrast, southern red-backed vole numbers were decreased for 2-3 years following burning (Sullivan and Boateng 1996). The studies cited above found stronger support for positive prescribed fire impacts on *P. maniculatus* abundances than reported by Monroe and Converse (2006) in Sierra Nevada mixed conifer forest, California. These authors said that the tendency for *P. maniculatus* densities to be greater on burned areas does not necessarily indicate that burned habitat is optimal for *P. maniculatus*, and may reflect dispersal to marginal sink habitat. In California oak woodland, the population of *N. fuscipes* increased from 1993 to 1997, and then decreased steadily after prescribed burning (Lee and Tietje 2005). *P. leucopus* increased population size in treatment sites (thinning, prescribed fire, and thinning + prescribed burning), but this increase was higher when the two types of treatments were combined in the southern Appalachian hardwood forest in North Carolina (Greenberg et al. 2006). A comparison between three sites (brush, prairie, and savanna) burned frequently during 15 years and three forested sites unburned during 35 years in western Wisconsin were carried out to study small mammal's abundance in these sites. *P. leucopus* and *Clethrionomys gapperi* (red-backed vole) were more common in the unburned forest, and *P. maniculatus* and *Spermophilus tridecemlineatus* (13-lined ground squirrel) were more abundant in the prairie created and maintained by fire. Burning the forest did not significantly reduce the number of mice present (Beck and Vogl 1972). In the first year after burning in the California grassland and chaparral, populations of *Chaetodipus californicus* (California pocket mouse), *Peromyscus californicus* (California mouse), and

Dipodomys agilis (agile kangaroo rat) were either unchanged or greater on burned than in unburned sites. However, populations of *R. humulis*, *Peromyscus boylii* (brush mouse), and *Neotoma spp*. (woodrat) decreased or disappeared in the burned sites (Wirtz 1977). Populations of *S. hispidus* in Arizona grassland were greatly reduced by a summer fire, while populations of *Perognathus hispidus* (seed-eating pocket mice) and *Dipodomys merriami* (kangaroo rats) increased. This difference was explained by the food habits of the species. *S. hispidus* fed on green vegetation that decreased after fire. The heteromyid rodents fed on seeds, which increased after fire due to the invasion of weedy forbs (Bock and Bock 1978). Even though there was no evidence that prescribed fire killed any small mammal, it negatively impacted *Mus musculus* (house mice), *P. maniculatus*, and *Microtus pennsylvanicus* (meadow voles) in Oxford, Ohio. *M. musculus* and *M. pennsylvanicus* disappeared from the burned site, while the three species prevailed in the control site (Crowned and Barret 1979). Populations of *Spermophilus* spp. (ground squirrels) and *Thonomys* spp. (pocket gophers) generally increase after fire (Ream 1981). Only *Dipodomy agilis* out of five species of rodents increased in abundance after prescribed burning in a coastal sage scrub in southern California (Prise and Waser 1984). Six small mammal species were not eliminated, and they did not increase in numbers in the months following prescribed burning in the California Chaparral (Lawrence 1966). A year after a prescribed burning in conifer woodland with shrubby understory in California, the abundance of small mammals was almost three times greater on unburned than burned sites, even though species composition did not vary significantly between burned and unburned sites (Blankenship 1982). The populations of the small mammal community (11 species) in the unburned sagebrush in Burro Hill, Wyoming, varied little before burning, the populations were at low level following burning, and populations approached control values three years after burning (McGee 1982). The impact of fire on small mammal communities in the central Appalachians on Pennsylvania was transitory, and the differences in small mammal abundance between unburned and burned sites disappeared within eight months after fire (Kirkland et al. 1996). Also, there were not significant differences in small mammals' mean total captured efficiency between treatment and control sites in southern Appalachian, North Carolina (Ford et al. 1999). Reduction of shrubs and woody debris by thinning with overly frequent prescribed fire may reduce small mammal densities (Converse et al. 2006). In summary, small mammal species' responses to prescribed burning vary greatly within the same geographic area and among states in the United States.

Habitat selection: immigration, emigration, and returning to burned areas

In general, prescribed burning affects small mammals mainly through the way it affects their habitats. Direct effects such as injury, mortality, and movement (immigration and emigration) might be the short-term population responses. Indirect effects through habitat alteration could influence long-term responses such as feeding, movement, reproduction, and availability of refugia (Smith 2000). In both circumstances, immigration and emigration play an important role in population demography, food availability, reproduction, and re-colonization of the burned areas. Immigration might occur because burned areas attract small mammals; however, emigration could also take place because there is insufficient food and cover in the burned area. Characteristics of an animal species such as mobility and particular food and cover requirements will determine its ability to re-invade a burned site (Whelan 1995). The length of time before these species return to burned sites depends on how much fire altered the habitat structure and food supply (Smith 2000). The last three sentences summarize and explain what *P. floridanus* and *S. hispidus* experienced at CKSSR.

Although I did not trap new individuals in burned sites following prescribed burning, there were data that showed possible immigration to the burned area. A total of 27 new adult individuals were trapped between the 5th and the 7th trapping sessions in wetlands corresponding to 3 *P. floridanus*, 20 *S. hispidus*, 3 *P. gossypinus*, and one *O. nuttalli*. These individuals were probably attracted by the burned area and moved to wetlands looking for refugia. Immigration to burned areas immediately after prescribed burning has been cited in the literature. Odors from burned areas might stimulate immigration of *P. maniculatus* from suboptimal habitats (Kaufman et al. 1988b). The high reproductive potential of *P. maniculatus* and *R. megalotis* populations in Kansas tallgrass prairie enables them to increase rapidly in favorable environments and disperse readily into recently burned areas (Kaufman et al. 1988b). The number of resident individuals of *P. floridanus* in burned areas of Ordway-Swisher Preserve was higher than in the unburned ones. *P. maniculatus* invaded a burned area after a heavy rain in California (Tevis 1956). Also, *P. maniculatus* invaded burned areas immediately after prescribed fire in jack pine in northeastern Minnesota (Ahlgreen 1966).

The lack of prescribed burning effect on *P. floridanus* and *S. hispidus* was probably because they used the vegetation surrounding wetlands as refugia. Statistically, I could not draw any conclusion, but practically mice may have moved to wetlands looking for cover and food. If mice did not have the wetlands near the treatment sites, probably they have had to move farther until finding another wetland or unburned site. *P. floridanus* in the current study moved out of burned areas. In contrast, Jones (1990) reported that all individuals of *P. floridanus* except one did not leave burned areas after prescribed burning in Ordway-Swisher Preserve. This is probably due a patchy burn in the Ordway sandhills in comparison with a more continuous burn in the CKSSR scrub. A similar result was found by Arata (1959) near Gainesville, Florida. The number of captured individuals of *P. floridanus* and *P. polionotus* remained at pre-burn levels in burned and unburned sites, whereas *S. hispidus* moved from the burned to the un-burned site following prescribed fire. However, Ream (1981) reported that red squirrel, voles, northern flying squirrel, and showshoe hare emigrated from recent burned areas. Of 25 species common in chaparral brushlands, Townsend's chipmunk and dusly-footed woodrat were not abundant in recently burned areas (Biswell 1989). This suggest that species such as *P. floridanus* found in the same geographic area might respond differently to prescribed burning depending on the habitat type and how much fire altered it. Also, other species always emigrate from burned sites, and most of the time they return to the same burned sites they moved away from months ago.

P. floridanus and *S. hispidus* returned to the burned sites in CKSSR after at least 11 months. This amount of time appears to be a long time for a mouse to live and survive. Out of 50 *P. floridanus* (20 individuals) and *S. hispidus*s (30 individuals) marked between the 5th and the 7th trapping sessions, 33 (66%) individuals (nine *P. floridanus* and 24 *S. hispidus*s) were recaptured again in scrub in the 8th trapping session. The survival curve for both species was high (Figures 4-6 and 4-8), and the amount of time involved between the 5th and the 8th trapping session was 373 days for 5C and 403 days for 2M. Are *P. floridanus* and *S. hispidus* long lived species? Jones (1990), at Smith Lake sandhill in Ordway Swisher Preserve, found that 8.6% of all marked mice (225 individuals) were present for 360 days or more. Of these mice, half were females first marked as juveniles, and most of the males were first marked as subadults and adults. The longevity records were 649 days for females and 920 days for males. Layne (1974) reported two *S. hispidus* females originally trapped as subadults were recorded on the study area during the entire 14-month period. A juvenile female and an adult and a juvenile male were first captured in October 1960 and they were recaptured 10 months later in July 1961.

P. floridanus and *S. hispidus* in CKSSR returned to the burned sites after they found cover and food to live there. It is surprising that both species returned approximately at least 11 months later. But, it is not as surprising when we consider the phenology of the plant species. Accorns of Myrtle oak, Chapman oak, and sand live oak developed surrounding wetlands in September, and production ended in December 2005. Thus, *P. floridanus* and *S. hispidus* had food to stay in the vegetation surrounding wetlands. In the burned sites 5C and 2M, the percentage of shrub cover at 11 months after prescribed burning was not quantified, but at 12 months was 72 % and 105%, respectively. This percent cover was mainly composed of *Quercus myrtifolia, Quercus geminata, Lyonia ferruginea, Lyonia lucida,* and *Quercus chapmanii* (see Tables 3-7 and 3-11). Neither flowering season nor fruiting season was synchronous in CKSSR within and among species, but *Vaccinium myrsinites* developed flowers in March and fruits in April 2006. *Gaylussacia nana* and *Serenoa repens* developed flowers in April, but *G. nana* had fruits in May and *Serenoa repens* in June 2006. Other species such as *L. ferruginea*, *L. lucida*, *Ilex glabra*, and *Brevaria racemosa* flowered in April-May and fruited in June-July 2006. Therefore, *P. floridanus* and *S. hispidus* returned to scrubs after plant species offered cover and food.

The relationship between the amount of cover and mice returning to burned areas has been reported in the literature. Arata (1959) indicated that *S. hispidus* did not return to longleaf/turkey oak habitat in north-central Florida at 5 months after a burn. However, he stated that the burned area was recolonized by *S. hispidus* within 6 months following the burn. Layne (1974) was the first study in Florida to report that the return of *S. hispidus* to a burned area appeared to be correlated with redevelopment of the ground cover in slash/longleaf pine habitat in north-central Florida. Ahlgren (1966) showed that the southern red-backed vole numbers decreased for 2-3 years in Minnesota following prescribed burning until recovery in the ground story vegetation occurred. McGee (1982) reported that the populations of the small mammal community (11 species) in the unburned sagebrush in Burro Hill, Wyoming, varied little before burning, the populations were at low level following burning, and populations approached control values three years after burning when total cover of the understory was near unburned levels. West (1982) indicated that northern red-backed voles avoided a burned area in black spruce for one year and established a resident population in post fire year 4, which it was the first year of berry production in central Alaska. Geluso et al. (1986) found that voles survived a prescribed burning in Nebraska grassland and left the burned areas until a new litter layer had accumulated about two growing seasons later. Possible reasons for emigration included decreased protection from predators and decreased food availability. Kirkland et al. (1996) showed that the impact of fire on small mammal communities in the central Appalachians on Pennsylvania was transitory, and the differences in small mammal abundance between unburned and burned sites disappeared within eight months after fire. This rapid recovery of small mammal populations was explained by the fast re-growth of ground cover within the study area, particularly of blueberry. Sullivan and Boateng (1996) reported that southern red-backed vole numbers decreased for 2-3 years in British Columbia following prescribed burning until recovery in the ground story vegetation took place. Schwilk and Keeley (1998) carried out a patchy burn in a California chaparral and coastal sage scrub. These refugia allow small mammals colonize severely burned sites during the first six months after prescribed fire. Ford et al. (1999) also used the link between small mammals and re-growth of the vegetation as the explanation of the population recovery in the study conducted in Southern Appalachian, North Carolina.

In summary, immediately after fire, some species are attracted to burned areas and immigrate into them. Other species emigrate due to insuficient food and cover in the burned area. The length of time before species return depends on how much fire altered the habitat structure and food supply.

Table 4-1. Number of captured individuals in the scrubby flatwoods and in the vegetation surrounding wetlands per trapping session in treatment and control sites in Cedar Key Scrub State Reserve.

	Trapping	*Podomys*			*Peromyscus*			*Ochrotomys*			*Sigmodon*			Total		
	Session	S	W	T	S	W	T	S	W	T	S	W	T	S	W	T
Treatment	1	4	3	7	2	1	3	7	1	8	1	0	1	14	5	19
	2	3	3	6	3	0	3	7	2	9	1	1	2	14	6	20
	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	4	9	4	13	5	2	7	1	0	1	11	2	13	26	8	34
	5	0	16	16	0	4	4	0	1	1	0	24	24	0	45	45
	6	1	15	16	1	6	7	0	0	0	0	23	23	2	44	46
	7	3	13	16	1	6	7	0	1	1	0	26	26	4	46	50
	8	21	0	21	2	2	4	0	0	0	30	4	34	53	6	59
			Total	95		Total	35		Total	20		Total	123	113	160	273
Control	1	7	3	10	0	0	0	11	0	11	1	0	1	19	3	22
	2	4	4	8	0	0	0	8	0	8	2	0	2	14	4	18
	3	8	1	9	0	0	0	0	0	0	1	0	1	9	1	10
	4	8	0	8	5	1	6	0	0	0	2	0	2	15	1	16
	5	6	0	6	6	0	6	1	0	1	9	0	9	22	0	22
	6	6	0	6	4	1	5	1	0	1	9	0	9	20	1	21
	7	7	0	7	3	0	3	0	0	0	11	0	11	21	0	21
	8	6	0	6	4	0	4	0	0	0	9	4	13	19	4	23
			Total	60		Total	24		Total	21		Total	48	139	14	153

Codes: S = scrubs. W = wetlands. T = Total.

Table 4-2. The Goodness of Fit test for the general model phi(g*t) p(g*t) for *Podomys floridanus* and *Sigmodon hispidus*.

Species	Bootstrap GOF Method	Observe	Mean	c-hat	GOF Method	Chi-square	df	c-hat
Podomys	Deviance method	12.5835	11.0510	1.1387	Release	0.9167	4	0.2292
	c-hat method	0.0000	0.0000	0.0000				
Sigmodon	Deviance method	24.4433	15.5750	1.5694	Release	5.6026	9	0.6225
	c-hat method	0.0000	0.0000	0.0000				

Table 4-3. Result browser of the fitted candidate model set of 16 models for *Podomys floridanus* after adjusting to c-hat = 1.1387 in treatment and control sites in Cedar Key Scrub State Reserve.

Models	QAICc	Delta QAICc	QAICc weight	Model likelihood	#Par	QDeviance
{Phi(t) p(.) PIM}	123.822	0.00	0.73160	1.0000	8	18.930
{Phi(t) p(g) PIM}	125.949	2.13	0.25265	0.3453	9	18.741
{Phi(t) p(t) PIM}	131.653	7.83	0.01458	0.0199	13	14.778
{Phi(g*t) p(.) PIM}	137.206	13.38	0.00091	0.0012	15	15.239
{Phi(g*t) p(g) PIM}	139.728	15.91	0.00026	0.0004	16	15.145
{Phi(g*t) p(t) PIM}	149.739	25.92	0.00000	0.0000	21	11.340
{Phi(t) p(g*t) PIM}	150.755	26.93	0.00000	0.0000	21	12.356
{Phi(.) p(t) PIM}	153.549	29.73	0.00000	0.0000	8	48.657
{Phi(g) p(t) PIM}	153.681	29.86	0.00000	0.0000	9	46.474
{Phi(.) p(.) PIM}	158.311	34.49	0.00000	0.0000	2	66.533
{Phi(g) p(.) PIM}	158.629	34.81	0.00000	0.0000	3	64.753
{Phi(.) p(g) PIM}	160.259	36.44	0.00000	0.0000	3	66.383
{Phi(g) p(g) PIM}	160.716	36.89	0.00000	0.0000	4	64.708
{Phi(g) p(g*t) PIM}	165.704	41.88	0.00000	0.0000	16	41.121
{Phi(.) p(g*t) PIM}	166.047	42.23	0.00000	0.0000	15	44.079
{Phi(g*t) p(g*t) PIM}	171.137	47.31	0.00000	0.0000	28	11.051

Table 4-4. Set of 25 models after adding phi(Flood + Fire) in combination with p constant (.), time dependent (t), and group dependent (g) in the analysis Flood and Fire effect on survival probabilities of *Podomys floridanus* in treatment and control sites in Cedar Key Scrub State Reserve.

Models	QAICc	Delta QAICc	QAICc weight	Model likelihood	#Par	QDeviance
{Phi(Flood+Fire) p(.)}	121.892	0.00	0.29916	1.0000	4	25.885
{Phi(Flood+Fire) p(t)}	122.950	1.06	0.17630	0.5893	10	13.387
{Phi(Flood) p(t)}	123.393	1.50	0.14128	0.4723	9	16.185
{Phi(Flood+Fire) p(g)}	123.668	1.78	0.12308	0.4114	5	25.494
{Phi(t) p(.) PIM}	123.822	1.93	0.11397	0.3810	8	18.930
{Phi(Flood) p(.)}	124.720	2.83	0.07274	0.2432	3	30.845
{Phi(t) p(g) PIM}	125.949	4.06	0.03936	0.1316	9	18.741
{Phi(Flood) p(g)}	126.384	4.49	0.03166	0.1058	4	30.377
{Phi(t) p(t) PIM}	131.653	9.76	0.00227	0.0076	13	14.778
{Phi(g*t) p(.) PIM}	137.206	15.31	0.00014	0.0005	15	15.239
{Phi(g*t) p(g) PIM}	139.728	17.84	0.00004	0.0001	16	15.145
{Phi(Fire) p(t)}	148.336	26.44	0.00000	0.0000	9	41.129
{Phi(.) p(g*t) PIM}	148.972	27.08	0.00000	0.0000	8	44.079
{Phi(g*t) p(t) PIM}	149.739	27.85	0.00000	0.0000	21	11.340
{Phi(t) p(g*t) PIM}	150.755	28.86	0.00000	0.0000	21	12.356
{Phi(Fire) p(.)}	151.999	30.11	0.00000	0.0000	3	58.123
{Phi(.) p(t) PIM}	153.549	31.66	0.00000	0.0000	8	48.657
{Phi(g) p(t) PIM}	153.681	31.79	0.00000	0.0000	9	46.474
{Phi(Fire) p(g)}	153.973	32.08	0.00000	0.0000	4	57.966
{Phi(.) p(.) PIM}	158.311	36.42	0.00000	0.0000	2	66.533
{Phi(g) p(.) PIM}	158.629	36.74	0.00000	0.0000	3	64.753
{Phi(.) p(g) PIM}	160.259	38.37	0.00000	0.0000	3	66.383
{Phi(g) p(g) PIM}	160.716	38.82	0.00000	0.0000	4	64.708
{Phi(g) p(g*t) PIM}	165.704	43.81	0.00000	0.0000	16	41.121
{Phi(g*t) p(g*t) PIM}	171.137	49.24	0.00000	0.0000	28	11.051

Table 4-5. Estimated survival (phi) and recapture (p) parameters by using model phi(Flood+ Fire) p(.) in the analysis Flood and Fired effect on survival probabilities of *Podomys floridanus* in treatment and control sites in Cedar Key Scrub State Reserve. Confidence interval = 95%.

Phi		Estimate	Standard error	Lower CI	Upper CI
1:Phi		0.7870664	0.0507500	0.6712425	0.8699882
2:Phi		0.0000000	0.0000000	0.0000000	0.0000000
3:Phi		0.7870664	0.0507500	0.6712425	0.8699882
4:Phi	Treatment	1.0000000	0.0000000	1.0000000	1.0000000
5:Phi		0.7870664	0.0507500	0.6712425	0.8699882
6:Phi		0.7870664	0.0507500	0.6712425	0.8699882
7:Phi		0.7870664	0.0507500	0.6712425	0.8699882
8:Phi		0.7870664	0.0507500	0.6712425	0.8699882
9:Phi		0.0000000	0.0000000	0.0000000	0.0000000
10:Phi		0.7870664	0.0507500	0.6712425	0.8699882
11:Phi	Control	0.7870664	0.0507500	0.6712425	0.8699882
12:Phi		0.7870664	0.0507500	0.6712425	0.8699882
13:Phi		0.7870664	0.0507500	0.6712425	0.8699882
14:Phi		0.7870664	0.0507500	0.6712425	0.8699882
15:p	Recapture	0.9430430	0.0398357	0.7946523	0.9860803

Table 4-6. Set of 28 models after adding phi(Flood + Fire) in combination with p(Flood + Fire), p(Flood), and p(Fire) in the analysis Flood and Fire effect on survival probabilities of *Podomys floridanus* in treatment and control sites in Cedar Key Scrub State Reserve.

Models	QAICc	Delta QAICc	QAICc weight	Model likelihood	#Par	QDeviance
{Phi(Flood+Fire) p(.)}	121.892	0.00	0.24033	1.0000	4	25.885
{Phi(Flood+Fire) p(t)}	122.950	1.06	0.14164	0.5894	10	13.387
{Phi(Flood) p(t)}	123.393	1.50	0.11350	0.4723	9	16.185
{Phi(Flood+Fire) p(g)}	123.668	1.78	0.09888	0.4114	5	25.494
{Phi(t) p(.) PIM}	123.822	1.93	0.09156	0.3810	8	18.930
{Phi(Flood + Fire) p(Fire)}	123.935	2.04	0.08652	0.3600	5	25.761
{Phi(Flood + Fire) p(Flood)}	124.059	2.17	0.08135	0.3385	5	25.885
{Phi(Flood) p(.)}	124.720	2.83	0.05843	0.2431	3	30.845
{Phi(t) p(g) PIM}	125.949	4.06	0.03162	0.1316	9	18.741
{Phi(Flood + Fire) p(Flood + Fire)}	126.138	4.25	0.02877	0.1197	6	25.761
{Phi(Flood) p(g)}	126.384	4.49	0.02543	0.1058	4	30.377
{Phi(t) p(t) PIM}	131.653	9.76	0.00182	0.0076	13	14.778
{Phi(g*t) p(.) PIM}	137.206	15.31	0.00011	0.0005	15	15.239
{Phi(g*t) p(g) PIM}	139.728	17.84	0.00003	0.0001	16	15.145
{Phi(Fire) p(t)}	148.336	26.44	0.00000	0.0000	9	41.129
{Phi(.) p(g*t) PIM}	148.972	27.08	0.00000	0.0000	8	44.079
{Phi(g*t) p(t) PIM}	149.739	27.85	0.00000	0.0000	21	11.340
{Phi(t) p(g*t) PIM}	150.755	28.86	0.00000	0.0000	21	12.356
{Phi(Fire) p(.)}	151.999	30.11	0.00000	0.0000	3	58.123
{Phi(.) p(t) PIM}	153.549	31.66	0.00000	0.0000	8	48.657
{Phi(g) p(t) PIM}	153.681	31.79	0.00000	0.0000	9	46.474
{Phi(Fire) p(g)}	153.973	32.08	0.00000	0.0000	4	57.966
{Phi(.) p(.) PIM}	158.311	36.42	0.00000	0.0000	2	66.533
{Phi(g) p(.) PIM}	158.629	36.74	0.00000	0.0000	3	64.753
{Phi(.) p(g) PIM}	160.259	38.37	0.00000	0.0000	3	66.383
{Phi(g) p(g) PIM}	160.716	38.82	0.00000	0.0000	4	64.708
{Phi(g) p(g*t) PIM}	165.704	43.81	0.00000	0.0000	16	41.121
{Phi(g*t) p(g*t) PIM}	171.137	49.24	0.00000	0.0000	28	11.051

Table 4-7. Estimated β parameters from models phi(Flood+ Fire) p(.) in the analysis Flood and Fired effect on the survival probabilities of *Podomys floridanus* in treatment and control sites in Cedar Key Scrub State Reserve. Confidence interval = 95%.

Model	Parameter	β	Standard error	Lower CI	Upper CI
	Interception	1.3073323	0.3028174	0.7138103	1.9008543
phi(Flood+Fire) p(.)	Flood	-22.5146010	0.0000000	-22.5146010	-22.5146010
	Fire	30.5618730	0.0000000	30.5618730	30.5618730
	p	2.8068159	0.7416407	1.3532002	4.2604316

Table 4-8. Estimated survival parameters (phi) by using model averaging for the set of 28 models in the analysis Flood and Fire effect on the survival probabilities of the *Podomys floridanus* in treatment and control sites in Cedar Key Scrub State Reserve. C-hat = 1.1387; 95% confidence interval.

	Phi	Estimate	Standard error	Lower CI	Upper CI
	1:phi	0.7847947	16.2037846	0.0000000	1.0000000
	2:phi	0.0000000	0.0000055	-0.0000108	0.0000108
Average	3:phi	0.8269939	20.3813127	0.0000000	1.0000000
Treatment	4:phi	0.9677876	0.0613967	0.3875942	0.9992993
	5:phi	0.8251911	0.0687758	0.6496347	0.9231836
	6:phi	0.8247459	0.0694662	0.6472239	0.9234956
	7:phi	0.7939472	49.0490509	0.0000000	1.0000000
	1:phi	0.7847475	16.2037852	0.0000000	1.0000000
	2:phi	0.0000000	0.0000055	-0.0000108	0.0000108
Average	3:phi	0.8270833	0.0763053	0.6269967	0.9315558
Control	4:phi	0.8353632	0.0781662	0.6248433	0.9392373
	5:phi	0.8251870	0.0687882	0.6495939	0.9231923
	6:phi	0.8247389	0.0694835	0.6471662	0.9235066
	7:phi	0.7939708	49.0490508	0.0000000	1.0000000

Table 4-9. Result browser of the candidate model set of 16 models for *Sigmodon hispidus* fitted and adjusted to c-hat = 1.5694 in treatment and control sites in Cedar Key Scrub State Reserve.

Models	QAICc	Delta QAICc	QAICc weight	Model likelihood	#Par	QDeviance
{Phi(t) p(.) PIM}	86.170	0.00	0.69776	1.0000	8	17.575
{Phi(t) p(g) PIM}	88.159	1.99	0.25817	0.3700	9	17.237
{Phi(.) p(.) PIM}	93.408	7.24	0.01871	0.0268	2	37.966
{Phi(g) p(.) PIM}	94.814	8.64	0.00926	0.0133	3	37.271
{Phi(.) p(g) PIM}	95.071	8.90	0.00815	0.0117	3	37.528
{Phi(g) p(g) PIM}	96.668	10.50	0.00367	0.0053	4	36.989
{Phi(t) p(t) PIM}	97.014	10.84	0.00308	0.0044	13	16.362
{Phi(.) p(t) PIM}	99.994	13.82	0.00070	0.0010	8	31.399
{Phi(g) p(t) PIM}	101.753	15.58	0.00029	0.0004	9	30.831
{Phi(g*t) p(.) PIM}	102.937	16.77	0.00016	0.0002	15	17.150
{Phi(g*t) p(g) PIM}	105.261	19.09	0.00005	0.0001	16	16.834
{Phi(.) p(g*t) PIM}	116.520	30.35	0.00000	0.0000	15	30.733
{Phi(g*t) p(t) PIM}	118.334	32.16	0.00000	0.0000	21	15.932
{Phi(t) p(g*t) PIM}	118.380	32.21	0.00000	0.0000	21	15.978
{Phi(g) p(g*t) PIM}	118.632	32.46	0.00000	0.0000	16	30.205
{Phi(g*t) p(g*t) PIM}	140.013	53.84	0.00000	0.0000	28	15.575

Table 4-10. Set of 22 models after adding phi(Flood + Fire) in combination with p (., g) to the candidate model set in the analysis Flood and Fire effect on survival probabilities of *Sigmodon hispidus* in treatment and control sites in Cedar Key Scrub State Reserve.

Models	QAICc	Delta QAICc	QAICc weight	#Par	Qdev.
{Phi(Flood) p(.)}	84.472	0.00	0.32506	3	26.930
{Phi(Flood + Fire) p(.)}	85.098	0.63	0.23770	4	25.419
{Phi(t) p(.) PIM}	86.170	1.70	0.13907	8	17.575
{Phi(Flood) p(g)}	86.220	1.75	0.13565	4	26.541
{Phi(Flood + Fire) p(g)}	86.838	2.37	0.09962	5	24.986
{Phi(t) p(g) PIM}	88.159	3.69	0.05145	9	17.237
{Phi(.) p(.) PIM}	93.408	8.94	0.00373	2	37.966
{Phi(Fire) p(.)}	94.797	10.33	0.00186	3	37.254
{Phi(g) p(.) PIM}	94.814	10.34	0.00185	3	37.271
{Phi(.) p(g) PIM}	95.071	10.60	0.00162	3	37.528
{Phi(Fire) p(g)}	96.463	11.99	0.00081	4	36.784
{Phi(g) p(g) PIM}	96.668	12.20	0.00073	4	36.989
{Phi(t) p(t) PIM}	97.014	12.54	0.00061	13	16.362
{Phi(.) p(t) PIM}	99.994	15.52	0.00014	8	31.399
{Phi(g) p(t) PIM}	101.753	17.28	0.00006	9	30.831
{Phi(g*t) p(.) PIM}	102.937	18.46	0.00003	15	17.150
{Phi(g*t) p(g) PIM}	105.261	20.79	0.00001	16	16.834
{Phi(.) p(g*t) PIM}	116.520	32.05	0.00000	15	30.733
{Phi(g*t) p(t) PIM}	118.334	33.86	0.00000	21	15.932
{Phi(t) p(g*t) PIM}	118.380	33.91	0.00000	21	15.978
{Phi(g) p(g*t) PIM}	118.632	34.16	0.00000	16	30.205
{Phi(g*t) p(g*t) PIM}	140.013	55.54	0.00000	28	15.575

Table 4-11. Estimated survival (phi) and recapture (p) parameters by using model phi(Flood) p(.) and phi (Flood + Fire) p(.) in the analysis Flood and Fired effect on survival probabilities of *Sigmodon hispidus* in treatment and control sites in Cedar Key Scrub State Reserve. Confidence interval = 95%.

Model	Parameter		Estimate	Standard error	Lower CI	Upper CI
phi(Flood) p(.)	1:phi	Treatment	0.9081981	0.0355371	0.8109827	0.9580029
	2:phi		0.0000000	0.0000523	0.0000000	0.7602520
	3:phi		0.9081981	0.0355371	0.8109827	0.9580029
	4:phi		0.9081981	0.0355371	0.8109827	0.9580029
	5:phi		0.9081981	0.0355371	0.8109827	0.9580029
	6:phi		0.9081981	0.0355371	0.8109827	0.9580029
	7:phi		0.9081981	0.0355371	0.8109827	0.9580029
	8:phi	Control	0.9081981	0.0355371	0.8109827	0.9580029
	9:phi		0.0000000	0.0000523	0.0000000	0.7602520
	10:phi		0.9081981	0.0355371	0.8109827	0.9580029
	11:phi		0.9081981	0.0355371	0.8109827	0.9580029
	12:phi		0.9081981	0.0355371	0.8109827	0.9580029
	13:phi		0.9081981	0.0355371	0.8109827	0.9580029
	14:phi		0.9081981	0.0355371	0.8109827	0.9580029
	15:p	Recapture	0.9295392	0.0352384	0.8212892	0.9742732
phi(Flood+Fire) p(.)	1:phi	Treatment	0.9275804	0.0353713	0.8202518	0.9729370
	2:phi		0.0000000	0.0000438	0.0000000	0.7246399
	3:phi		0.9275804	0.0353713	0.8202518	0.9729370
	4:phi		0.7736407	0.1472561	0.3967210	0.9467035
	5:phi		0.9275804	0.0353713	0.8202518	0.9729370
	6:phi		0.9275804	0.0353713	0.8202518	0.9729370
	7:phi		0.9275804	0.0353713	0.8202518	0.9729370
	8:phi	Control	0.9275804	0.0353713	0.8202518	0.9729370
	9:phi		0.0000000	0.0000438	0.0000000	0.7246399
	10:phi		0.9275804	0.0353713	0.8202518	0.9729370
	11:phi		0.9275804	0.0353713	0.8202518	0.9729370
	12:phi		0.9275804	0.0353713	0.8202518	0.9729370
	13:phi		0.9275804	0.0353713	0.8202518	0.9729370
	14:phi		0.9275804	0.0353713	0.8202518	0.9729370
	15:p	Recapture	0.9268919	0.0357070	0.8186314	0.9726868

Table 4-12. Estimated β parameters from models phi(Flood) p(.) and phi(Flood + Fire) p(.) in the analysis Flood and Fire effect on the survival probabilities of *Sigmodon hispidus* in treatment and control sites in Cedar Key Scrub State Reserve. Confidence interval = 95%.

Model	Parameter	β	Standard error	Lower CI	Upper CI
	Intersection	2.29	0.4262	1.4564	3.1272
phi(Flood) p(.)	Flood	-21.14	7999.2294	-15699.6274	15657.3518
	p	2.58	0.5380	1.5251	3.6342
	Intersection	2.55	0.5266	1.5181	3.5822
	Flood	-21.58	8080.7068	-15859.7679	-15816.7679
phi(Flood+Fire) p(.)	Fire	-1.32	0.9970	-3.2752	0.6329
	p	2.54	0.5269	1.5071	3.5727

Table 4-13. Estimated survival parameters (phi) by using model averaging for the set of 25 models in the analysis Flood and Fire effect on the survival probabilities of *Sigmodon hispidus* in treatment and control sites in Cedar Key Scrub State Reserve. C-hat = 1.5694; 95% confidence interval.

Site	Phi	Estimate	Standard error	Lower CI	Upper CI
	1:phi	0.9225670	0.0406832	0.7960088	0.9732462
	2:phi	0.0075199	0.0148244	0.0215359	0.0365758
	3:phi	0.8513742	0.1855291	0.2444639	0.9902356
Treatment	4:phi	0.8413499	0.1178554	0.4844399	0.9676693
	5:phi	0.9160235	0.0393425	0.8001250	0.9674515
	6:phi	0.9225670	0.0406832	0.7960088	0.9732462
	7:phi	0.9162478	0.0761627	0.6099526	0.9871024
	1:phi	0.9225064	0.0407450	0.7957258	0.9732474
	2:phi	0.0074594	0.0147077	0.0213677	0.0362864
	3:phi	0.8512919	0.1301067	0.4330015	0.9772273
Control	4:phi	0.9037845	0.0523791	0.7425754	0.9683423
	5:phi	0.9159611	0.0394113	0.7998231	0.9674598
	6:phi	0.9225064	0.0407450	0.7957258	0.9732474
	7:phi	0.9161860	0.0760357	0.6108419	0.9870342

Table 4-14. Predators captured before the 8th trapping session in treatment and control sites in Cedar Key Scrub State Reserve.

Predator	5C	2M	5A	5D
Raccoon	18	15	6	5
Grey Fox	2	3	1	1
Opossum	7	9	2	3

Table 4-15. Trapping effort carried out by several studies conducted on small mammals in Florida. Code: TN = trapping nights.

Author	Study Area	Species	Vegetation Type	TN
Jones (1990)	Ordway/Swisher Preserve	*Podomys floridanus*	Sandhill	33,000
Current study	Cedar Key Scrub Preserve	Several	Scrubby Flatwoods	29,340
Frank (1996)	Anastasia Island	*Peromyscus polionotus*	Dune	23,200
Fitzgerald (1990)	Myakka River State Park	Several	Dry Prairie	15,608
Layne (1974)	North Central Florida	Several	Scrubby Flatwoods	11,370
Franz et al. (1998)	Avon Park	*Podomys floridanus*	Scrub	8,576
Newman (1997)	Ordway/Swisher Preserve	*Podomys floridanus*	Sandhill	6,084
Depeu (2005)	3 study areas, central FL	*Podomys floridanus*	Scrub, scrubby flatwoods, others	4,458
Morgan (1998)	Cedar Key Scrub Preserve	Several	Scrubby Flatwoods	2,970
Arata (1959)	North Central Florida	Several	Longleaf Pine/Turkey Oak	1,950
Humphrey et al. (1985)	Ordway/Swisher Preserve	*Podomys floridanus*	Sandhill	1,114
Sasso and Gaines (2002)	Key Largo	Several	Hardwood Hammocks	66
Vogl (1973)	North Florida	Several	Hardwood forest	5

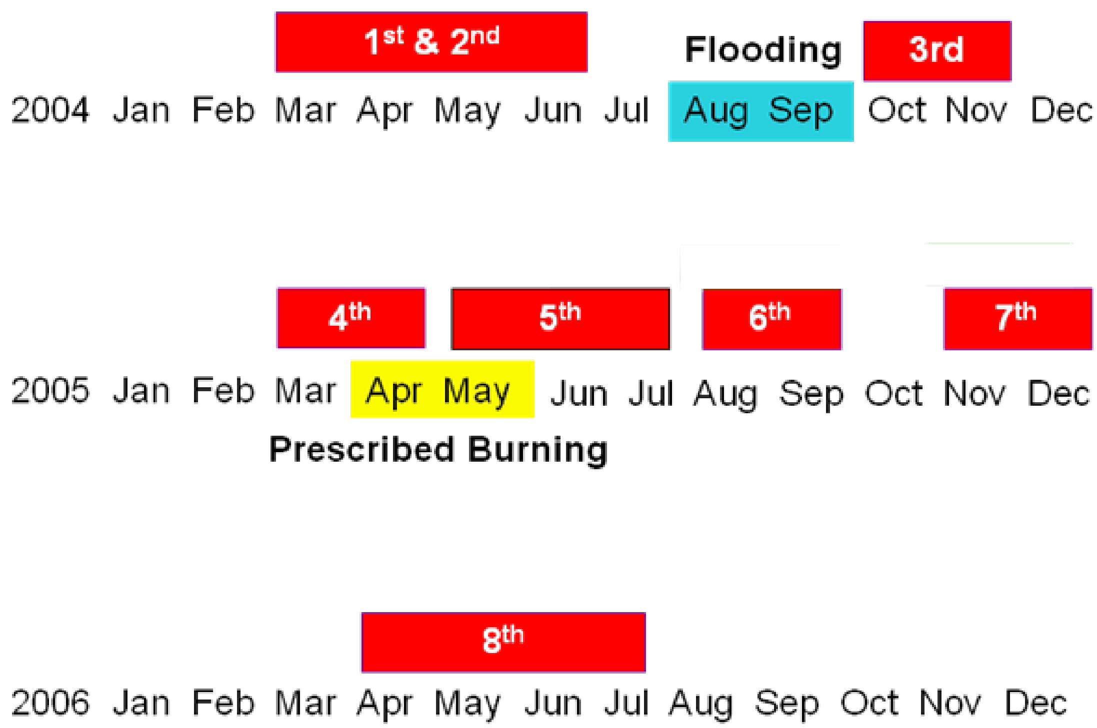


Figure 4-1. Trapping sessions (red blocks) carried out in Cedar Key Scrub State Reserve. Three hurricanes hit Cedar Key before the 3rd trapping session, and prescribed burning occurred before the 5th trapping session.

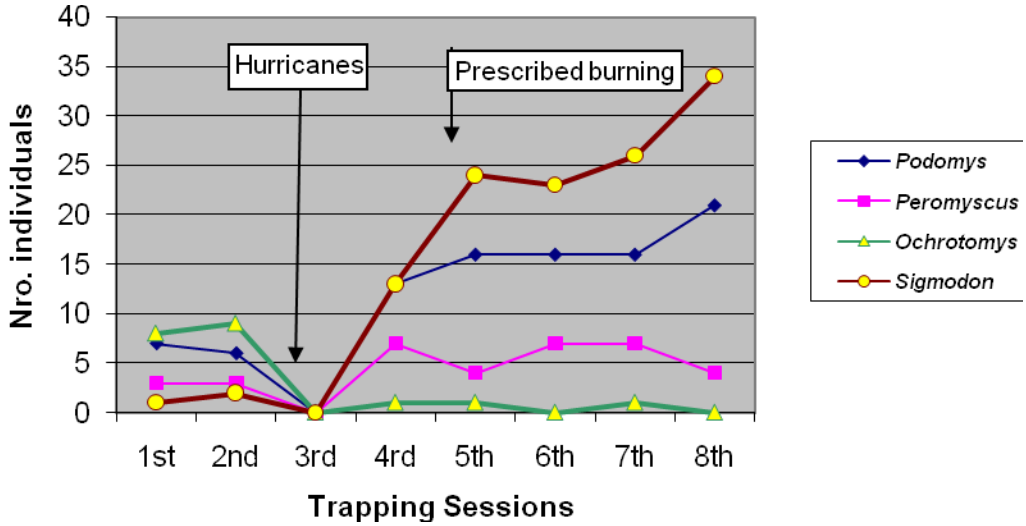


Figure 4-2. Number of captured individuals per species per trapping session in treatment sites 5C and 2M in Cedar Key Scrub State Reserve.

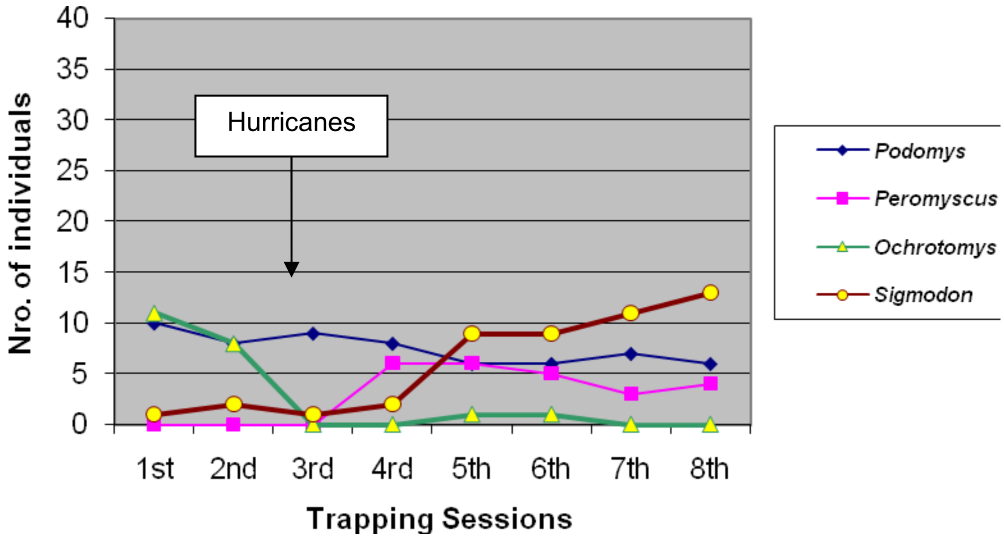


Figure 4-3. Number of captured individuals per species per trapping session in control sites 5A and 5D in Cedar Key Scrub State Reserve.

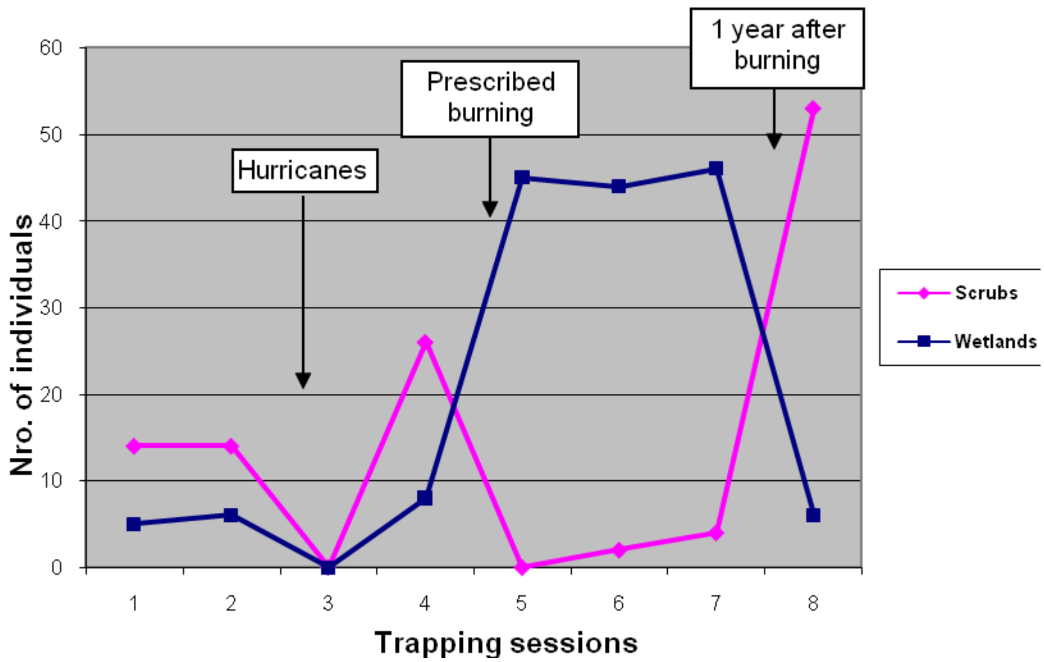


Figure 4-4. Number of captured individuals in scrubs and wetlands per trapping session in treatment sites 5C and 2M in Cedar Key Scrub State Reserve.

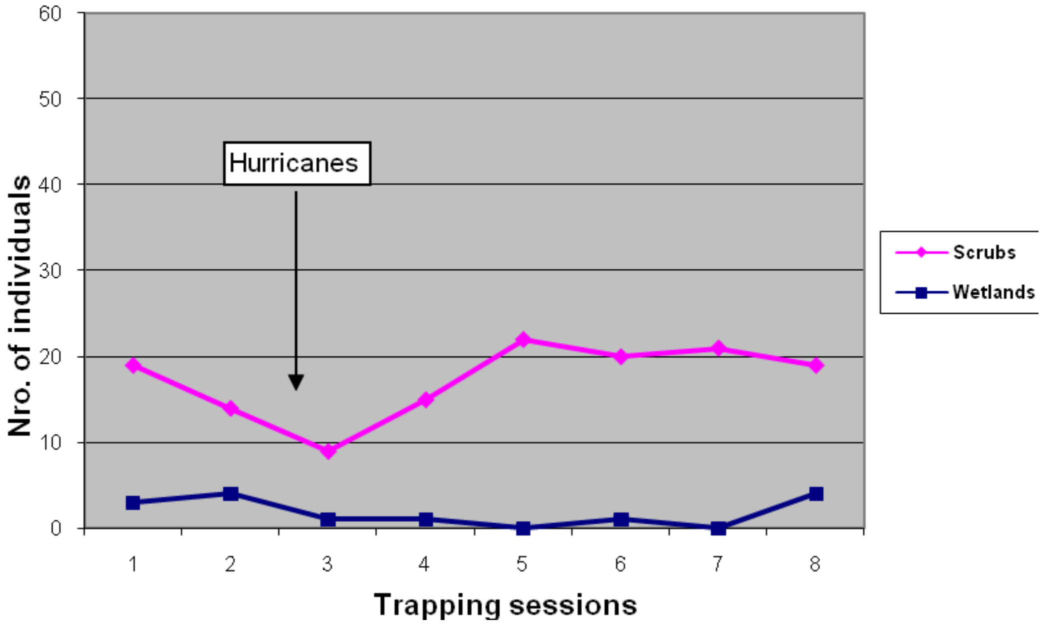


Figure 4-5. Number of captured individuals in scrubs and wetlands per trapping session in control sites 5A and 5D in Cedar Key Scrub State Reserve.

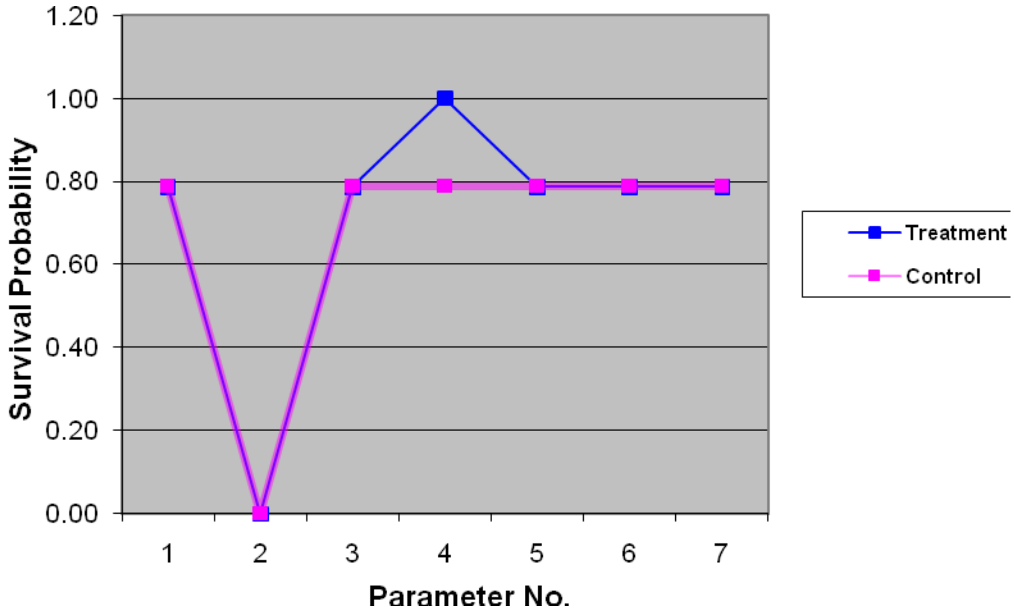


Figure 4-6. Survival probabilities of *Podomys floridanus* estimated by the model phi(Flood + Fire) p(.) in the set of 28 models in treatment and control sites in Cedar Key Scrub State Reserve (chat=1.1387).

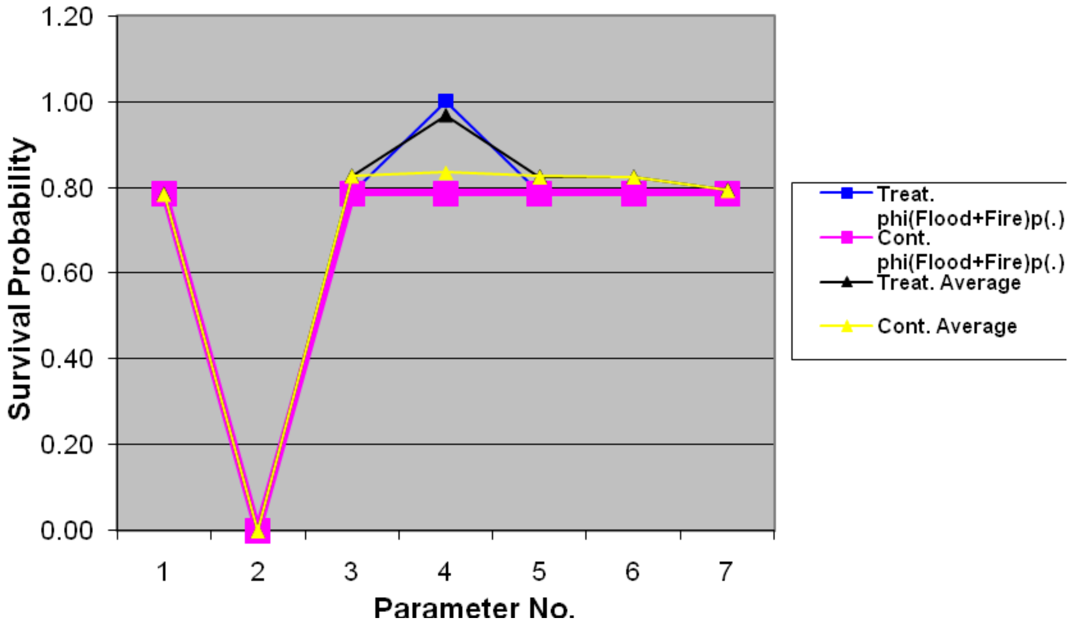


Figure 4-7. *Podomys floridanus's* survival probabilities calculated by the model phi(Flood + Fire) p(.) and by model averaging (28 models) in treatment and control sites in Cedar Key Scrub State Reserve (chat=1.1387).

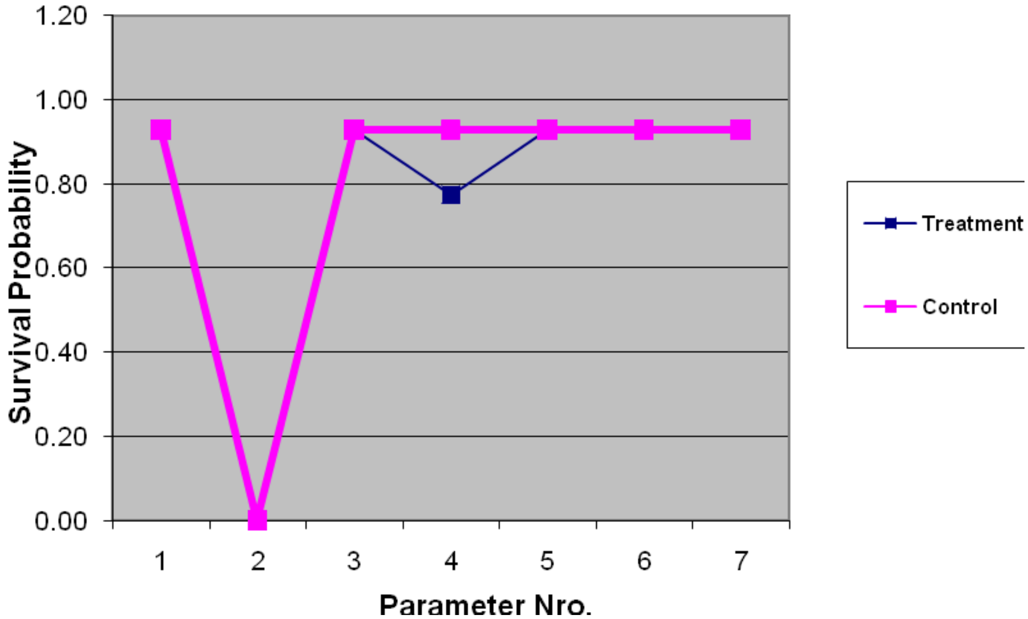


Figure 4-8. Survival probabilities of *Sigmodon hispidus* quantified by the model phi(Flood + Fire) p(.) in the set of 22 models in treatment and control sites in Cedar Key Scrub State Reserve (chat=1.5694).

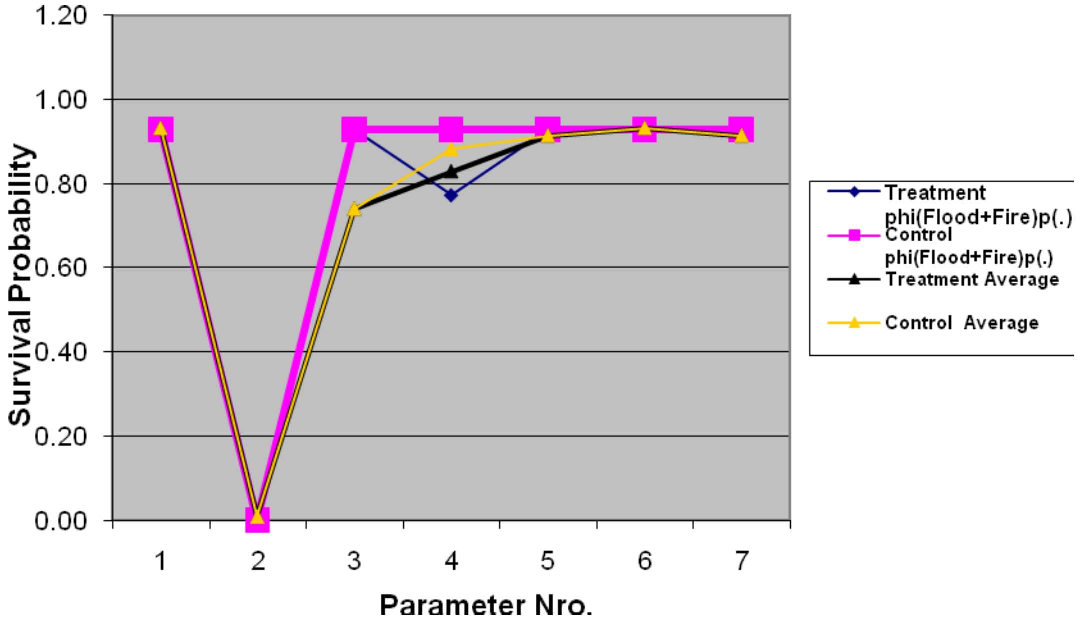


Figure 4-9. *Sigmodon hispidus*'s survival probabilities estimated by the model phi(Flood + Fire) p(.) and by model averaging in the set of 22 models in treatment and control sites in Cedar Key Scrub State Reserve (chat=1.5694).

CHAPTER 5
CONCLUSIONS AND RECOMMENDATIONS

Conclusions

Prescribed Burning Plan

The prescribed burning plan carried out in treatment sites 5C and 2M achieved the management and the research objectives. Burning reduced the vegetative mass on the ground more than 60%, and top-killed more than 75% of the woody vegetation. Wheather this management action in 5C and 2M will improve the habitat for scrub jays, needs to be evaluated following this study. Even though the prescribed burning plan was almost the same for each treatment site, fire behavior characteristics were not exactly the same. Rate of spread and flame lengths were similar, but fire intensity in 5C was lower and varied more than in 2M, which was more homogeneous across the site. The change in wind speed during the burn of 5C might be one factor responsible for this difference. Even though 5C and 2M sites did not have exactly the same fire behavior characteristics, the outcome of reducing and top killing the above ground vegetation was the same in both sites. This result made possible the comparison of plant responses between treatment and control sites.

Plant Species Responses to Prescribed Burning

A cluster analysis in combination with an ordination technique and a F-ratio test (with the respective multiple comparison test) was used to carry out a site analysis. Statistically, treatment and control sites in CKSSR were ecologically similar, and they were compared to determine prescribed burning effects.

Resprouting was the main way of surviving fire and recovery by the majority of the species in CKSSR. During the first 12 months in treatment sites, the recovery of the vegetation was so fast that bareground did not have values higher than 13%, and litter achieved more than 69% of the preburn level. Debris and vegetation height were the variables with the slowest recovery, having debris 32% and height 31% of the preburn value at 12 months.

The recovery process of herb and woody species was different. The Family Poaceae was dominant over other herb species and was the only taxon with preburn value. Only *Galactia elliottii* and *G. mollis* recuperated control values at 12 months postburn. *Solidago odora, Crotalaria rotundifolia*, and *Woodwardia virginica* were only recorded with low densities and cover during the postburn period and the other five species were sampled only in control sites. Prescribed burning created gaps that were temporarily used by *S. odora, C. rotundifolia*, and *W. virginica* and eventually filled by the growing woody vegetation.

The seven woody species with high densities, frequencies, cover, and importance values in treatment and control sites were: *Quercus myrtifolia, Serenoa repens, Q. geminata, Lyonia ferruginea, Lyonia lucida, Q. chapmanii, and Vaccinium myrsinites. Q. myrtifolia and S. repens* were the most dominant species. Rare species were: *Quercus nigra, Quercus sp., Ceratolia ericoides, Osmanthus americanus, Salix caroliniana, Smilax sp*, and *Opuntia humifusa*. In general, the seven dominant species recovered preburn values (except for height) and/or achieved at least a control value at 3 months with the exception of *V. myrsinites*. Also, there was a shift in dominance for *Q. chapmanii* and *L. ferruginea* through the 12 month period. Tree mortality was low in quadrats (7.5%) because oaks and ericaceous shrubs are adapted to fire and pine trees had very low densities, but pine tree mortality was high in grids (91.6%). The Detrended Correspondence Analysis indicated that woody species had structural and compositional changes during the first 3 months postburn, but there were more compositional than structural changes after that as one of the effects of prescribed burning. According to the Multi-response Permutation Procedure, the structural changes were significant, which means that prescribed burning had a significant effect on absolute densities on treatment sites between pre- and 12 months postburn. *V. myrsinites* was the first species to have flowers in March and fruits in April, which was an important factor for small mammals returning to burn sites.

Small Mammal Responses to Prescribed Burning

The combination of Flood and Fire effects affected the survival probability of the Florida mouse. Model phi(Flood + Fire) p(.) was the most parsimonious model. In this model, even though I cannot statistically conclude that Flood decreased survival from 0.8732 to 0.0000 in all sites because the β parameter was not estimable, practically and most likely, marked mice died during or after flooding. Also, although I cannot statistically state that prescribed fire increased survival (from 0.7871 to 1.0000) in treatment sites because the β parameter was not estimable, most likely prescribed fire forced the Florida mouse to emigrate from the scrubs to the vegetation surrounding wetlands, and mice were able to survive. Models in which Fire was involved alone did not have support in the data. Therefore, prescribed burning did not negatively affect the survival probability of the Florida mouse because the vegetation surrounding wetlands were used as refugia after prescribed burning for at least 11 months.

The covariate Flood and the additive effect of Flood and Fire influenced the survival probability of the cotton rat. Models phi(Flood) p(.) and phi(Flood + Fire) p(.) were the most parsimonious ones. Statistically, I cannot conclude that Flood in both models decreased survival from values such as 0.9082 and 0.9276 to 0.0000 between the 2nd and 3rd trapping sessions because the β parameters were not estimable. However, marked cotton rats probably died after flooding before the 3rd trapping session. Fire in model phi(Flood + Fire) p(.) did not significantly decrease survival from 0.9276 to 0.7736 between the 4th and the 5th trapping sessions because the β parameter was not significant. Hence, prescribed burning did not have a significant negative effect on the survival probability of the cotton rat because the majority of these rats emigrated from scrubs to the vegetation surrounding wetlands and used it as refugia for at least 11 months.

The number of individuals of Florida mice and cotton rats increased after prescribed burning because of immigration of new adult individuals in the vegetation surrounding wetlands. The cotton mouse maintained relatively low, but stable numbers of individuals, in treatment and control sites. Probably, competition for food and space did not allow immigrant cotton mice to establish in the vegetation surrounding wetlands. The golden mouse drastically declined in the number of individuals because of its arboreal life form and the combined effect of flooding and prescribed burning. These species have been reported with a similar or different reaction to prescribed fire at other localities in Florida and the United States. Therefore, these species do not have a specific pattern in their responses to prescribed burning.

Emigration to refugia occurred because there was no cover and food in the burned sites. Immigration of new adult individuals possibly took place because burned areas attracted rodents, but they had to move to the vegetation surrounding wetlands because of the lack of food and cover. The Florida mouse and the cotton rat were the only species that established in the wetlands, where they had acorns from September to December. Returning to the scrubs happened in or after March 2006, where the amount of cover was at least 70%, and only *V. myrsinites* were flowering. Fruit production started with *V. myrsinites* in April 2006. So, the re-colonization of the burned sites can be explained by the re-growth of the shrub cover and the production of fruits by *V. myrsinites*.

Recommendations

Prescribed burning studies in Florida are strongly needed. Regarding the amount of knowledge obtained from prescribed burning studies, Florida is behind in comparison with studies conducted in California, Arizona, Montana, and the Appalachian region. Arata (1959) wrote: “Considering the ecological importance and frequency of fire in the southeast, the lack of data on its effects on small mammals is surprising.” Jones (1990) also stated: ”Finally, I wish to emphasize how little is known about the effects of fires on small mammals….It is rather surprising; however, that with the prevalence of both prescribed burns and lightning strikes in the southeastern United States, there is still so little research on effects on non-game animals of the region.” It is even more surprising that Arata’s and Jone's statements are still valid in 2008.

The experimental design constraint of having the same fire behavior characteristics in treatment sites is extremely difficult to satisfy. However, an attempt must be made in order to try to achieve this important goal. In this regard, this dissertation recommends burning treatment sites in the same day when possible or on different dates not separated by more than one month.

The season of burning is critical for both plant species and wildlife. A prescribed fire too early in the season might kill flowering buds or developing flowers. This mortality could significantly reduce a full year of potential seeds. Sprouting shrubs may take some years to recover sufficiently to support flowering again (Whelan 1995). Also, it is critical to avoid prescribed burning during periods of reproduction in small mammals in order to reduce juvenile mortality and to increase the possibility of re-nesting in bird species. Spring fires may impact small mammal populations more than fires in other seasons because of limited mobility of young. The species with the most vulnerable young are small mammals, most of which also have high reproductive rates. If postfire habitat provides food and shelter for them, their populations recover rapidly. Following these ideas, land managers should limit the size of prescribed fires during peak of reproductive periods, which occur during February-March in south Florida and May-June in north Florida. Hence, according to the current information available for CKSSR, prescribed burning should be applied in April to May because the majority of the plant species have produced seeds and small mammals have reproduced.

Do not burn wetlands when they are the only refugia available, and burn them when other habitats might work as refugia. I suggest that the prescribed burning plan should consider refugia for wildlife. If the area to be burned does not have surrounded habitats not included in the prescribed burning plan, wetlands next to the area to be burned should not be included in the prescribed burning plan. These wetlands will be the only refugia available for wildlife. Otherwise, they should be burned because the vegetation will change to another type of habitat. Another possibility is to burn wetlands from one to 2 years out of the sequence with the scrub, after giving the scrub enough time to restablish cover and food production.

The prescribed burning plan should not include large areas to be burned in the same season. It would not be wise to include large extension of land because the prescribed fire plan might fail. Large extension of land could not be controlled, and the damage to both plants and wildlife may be irreversible. Land managers always make the assumption that wildlife, especially small mammals, will recover soon from adjacent habitats. Particularly, if we are talking about burning small plots, there is no doubt about it. But, if large areas are considered, the prescribed burning might drastically cause decline or even eliminate rare species. In addition, there is no guarantee that rare species would recover because in general prescribed burning plans do not have a monitoring program working at the same time to measure fire effects on vegetation and wildlife. In contrast, burning small plots makes the job easier, allows wildlife to recolonize burned sites faster, and makes the restoration goals feasible.

Mechanical treatments have already started to be used in CKSSR, but cautio should be considered. The rapid recovery to previous conditions occurs in scrub only when the sprouting ability of the dominant shrubs remains intact. Mechanical disturbances that remove roots and rhizomes of oaks, palmettos, and ericaceous shrubs cause long-lasting changes in structure and composition (Breininger and Schmalzer 1990, Schmalzer et al. 2003). Therefore, mechanical treatment should be inspected *in situ* to assure no damage of root and rhizome systems.

Fire return interval should be established based on the knowledge of plant species response to prescribed burning. This aspect is critical in any public land where restoration through prescribed burning is needed. Prescribed burning should not be applied in short intervals because it might occur at the beginning of seed production of obligate seeders such as *P. clausa* or *C. ericoides.* Also, it should not be applied in the long intervals because obligate-seeder species like herb species are lost because oaks and ericaceous shrubs storage large amounts of underground resources. For this reason, it is very important to monitor structural and compositional change after fire in order to determine when the community returns to conditions similar to stands without fire suppression. This criterion might be used in order to establish a fire return interval for scrubby flatwoods in CKSSR.

Currently, CKSSR is applying prescribed burning every 5 years. Fire-return interval for scrubby flatwoods is one every 5-60 yr (Abrahamson and Abrahamson 1984a, Abrahamson and Hartnett 1990). The research program carried out in Kennedy Space Center indicates that 5 years is the average time for communities without fire suppression to return to preburn conditions. Therefore, CKSSR is applying the fire return interval for the reserve according to the literature**.** However, CKSSR needs a researcher in charge of monitoring prescribed burning effects on plants and wildlife and to determine if 5 years is the right time for the Reserve. The recovery process in CKSSR is so fast (for the majority of the vegetation variables) that debris cover and vegetation height might be taken as criteria because they have the slowest recovery among all variables.

Future research should address microsite post-fire conditions, patchiness within burns, and seasonality of fire effects for specific ecosystems. Few studies about microclimatic conditions before and after prescribed burning have been conducted in Florida. A more comprehensive database is needed in order to have a better understanding of the changes in environmental variables after prescribed burning and their effects on small mammals. The importance of food, cover, and predation as important factors regulating post-fire small mammal populations require investigations. The responses of small mammal populations to post-fire vegetation changes, especially in relation to fire patchiness, are important aspects to take into consideration because they might have a strong impact on both the responses of plant population and of small mammals to prescribed burning. To improve long-term management for sustaining ecosystems, information is needed about the effects of fire on small mammals, at different seasons and under different conditions and over several decades.

APPENDIX A

TWO TAILS T TEST COMPARISON AMONG ENVIRONMENTAL VARIABLES AND FIRE BEHAVIOR CHARATERISTICS BETWEEN SITES 5C AND 2M. $\alpha = 0.05$

Variable	Mean	Variance	N	T Stat	P-value
Wind Speed 5C (Km/h)	3.2	0.81	9	-0.81	0.4336
Wind Speed 2M (Km/h)	3.6	1.30	8		
Air Temperature 5C (□C)	26.6	1.27	8	-3.10	0.0112
Air Temperature 2M (°C)	29.3	4.82	8		
Relative Humidity 5C (%)	48.5	23.43	8	-4.07	0.0016
Relative Humdity 2M (%)	61.8	61.36	8		
1h timelag 5C	6.5	2.29	20	-1.46	0.1553
1h timelag 2M	7.4	6.29	20		
10h timelag 5C	7.5	18.17	20	-0.60	0.5550
10 h timelag 2M	8.3	17.91	20		
100h timelag 5C	13.9	85.36	20	-0.29	0.7768
100h timelag 2M	14.8	97.57	20		
Live Herb 5C	59.2	279.03	20	0.63	0.5298
Live Herb 5C	55.9	218.57	20		
Live Woody 5C	60.6	109.44	20	-0.52	0.6038
Live Woody 2M	62.1	69.79	20		
Flame Length 5C	4.0	1.73	30	1.59	0.1171
Flame Length 2M	3.4	1.95	30		
Back fire 5C	1.7	0.24	30	1.09	0.3009
Back fire 2M	1.5	0.15	30		
Head fire 5C	6.2	2.82	30	2.11	0.0641
Head fire 2M	4.9	1.50	30		
Temperature - ground 5C (°C)	857.9	141510.46	30	-2.00	0.0506
Temperature - ground 2M (°C)	1019.0	52172.59	30		
Temperature - above ground 5C (°C)	600.4	190308.66	30	-2.94	0.0048
Temperature - above ground 2M (°C)	898.6	118459.62	30		

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BIOGRAPHICAL SKETCH

I was raised in Caracas, Venezuela, where I received the degree "Licenciado en Biología" at Central University of Venezuela in 1981. From 1981 to 1985, I completed research on population ecology, community ecology, and the behavior of snakes and small mammals. In addition, I carried out the first evaluation of human impacts in a national park in Venezuela. From 1985 to 1990, I worked for three projects sponsored by New York Zoological Society-The Wildlife Conservation Society. These included (a) behavioral ecology and conservation of the Family Cracidae (birds) in Venezuela, (b) human impacts on cracids populations in Venezuela, and (c) uses, preferences, and hunting impact on wildlife in Venezuela. I was responsible for the last two projects. Based on my results in relation to hunting pressure in national parks, an educational program was recommended to the Venezuelan government.

From 1990 to 1996, I coordinated an educational program for hunters in four national parks. I obtained a LASPAU-Fulbright scholarship in 1996 and began my master's program in the University of Florida in 1997. I graduated in December 1998 and started the PhD program in January 1999. The course work was completed in 2001 and presented the qualifying exams in February 2002. Unfortunately, I did not obtain funding until October 2003 after changing the original project. From October 2003 to August 2006, I carried out field work for my dissertation in Cedar Key Scrub State Reserve in Florida.

I have coordinated the CIRCA Operations Training Program in the University of Florida since fall 2000. I have trained 298 technology consultants for assisting professors, students, and staff in five computers labs. In addition, I trained all supervisors and training specialists during this time.

I have published a book, a section in another book, 16 scientific publications, 20 technical reports, six publications for nonscientific audience, and I have given 38 speeches in Venezuela and in international congresses.