EFFECTS OF LONGLEAF PINE MANAGEMENT PRACTICES ON THE

HERPETOFAUNA OF SOUTH ALABAMA

Ashley Elizabeth Rall

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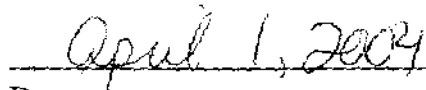Master of Science

Auburn, Alabama
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Ashley Elizabeth Rall

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THESIS ABSTRACT

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HERPETOFAUNA OF SOUTH ALABAMA

Ashley Elizabeth Rall
Master of Science, May 14, 2004
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Due to the suppression of natural fires in longleaf pine ecosystems, anthropogenic

management practices, including prescribed burns, thinning, and herbicide, are utilized in

order to sustain and regenerate this habitat. These practices may have positive or

negative effects on the herpetofauna. My data indicate a prolonged sampling period of

up to 21 years may be required in order to adequately quantify the effects of longleaf pine

management practices on species richness of amphibians and reptiles. Variation between

reptiles and amphibians and among years and sites play the dominant role in explaining

differences in species composition. Reptile captures rates were higher overall in

comparison to amphibian capture rates. Measures of evenness were significantly

different between taxa, with amphibians having greater evenness. We found no trend

towards specialist species inhabiting treated sites, although this lack of significance may

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occur due to an insufficient sampling period. Jaccard indices indicate a trend towards increased similarity among sites in as little as one year's time, regardless of treatment. Greater amounts of rainfall in 2003 most likely contributed to greater amphibian activity and reproduction. Due to preexisting site variation, treatment effects on the herpetofaunal community may have been masked in this study. However, analysis of three common species, *Bufo terrestris*, *Gastrophryne carolinensis* and *Scincella lateralis*, indicate that treatments improved habitat for specialist species and reduced it for resident species. My data indicate a strong need for long-term herpetofaunal studies within the longleaf pine ecosystem.
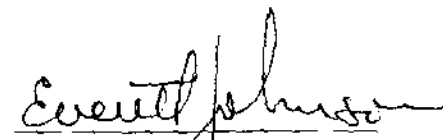
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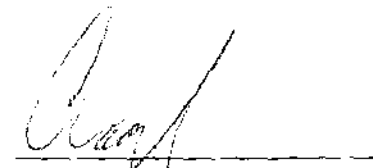
HERPETOFAUNA OF SOUTH ALABAMA

Except where reference is made to the work of others, the work described in this thesis is my own or was done in collaboration with my advisory committee. This thesis does not include proprietary or classified information.

_____
Ashley Elizabeth Rall

Certificate of Approval:

_____
Sharon Hermann
Visiting Assistant Professor
Biological Sciences

_____
Craig Guyer, Chair
Professor
Biological Sciences

_____
Everett Johnson
Director, SDWEC
School of Forestry and Wildlife
Sciences

_____
Stephen L. McFarland
Acting Dean
Graduate School

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BACKGROUND

Longleaf pine herpetofauna

Within the longleaf pine ecosystem, there are 73 species of amphibians and 95 species of reptiles whose geographic ranges overlap with longleaf and adjoining habitat (Guyer & Bailey 1993). Currently the Gopher Frog (*Rana capito*), Flatwoods Salamander (*Ambystoma cingulatum*), Striped Newt (*Notophthalmus perstriatus*), Florida Sand Skink (*Neoseps reynoldsi*), and Florida Scrub Lizard (*Sceloporus woodi*) are declining in numbers due to loss of natural longleaf pine habitat (Brennan et al. 1998). The Gopher Tortoise (*Gopherus polyphemus*) and Eastern Indigo Snake (*Drymarchon corais*) are listed as threatened under the Endangered Species Act, due in part to the suppression of fire (Brennan et al. 1998). With so many herpetofaunal species in danger of losing their native habitat, it is imperative to understand the impacts of longleaf pine management practices on the species richness and population biology of these organisms. In this study I examine the effects of four longleaf pine management practices on herpetofaunal richness and diversity in south Alabama. I also examine effects of these practices on the population biology of three of the most common species. From these data I infer how management practices affect local herpetofaunas and describe likely patterns of species resilience and persistence within anthropogenic-maintained longleaf pine ecosystems.

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Management treatments

Due to widening urban sprawl, natural management regimes for longleaf ecosystems are, in many instances, no longer applicable. Land managers have adapted several techniques in attempts to mimic historic natural processes like fire. Prescribed burning, mechanical thinning, and herbicide applications have all been applied to current longleaf stands, alone or in combinations, as management tools. Although fire is the historical disturbance that maintained longleaf ecosystem function and structure, land managers seek alternative methods of longleaf sustainability in order to avoid fire and smoke hazards associated with burning in proximity to human residences (Smith et al. 1997).

Fire

Fire contributes many positive attributes to longleaf pine ecosystem health. Fire releases nutrients in the form of carbon and ash, aids in seed germination, and affects soil temperatures and moisture (Brennan et al, 1998, Cavitt, 2000). Fire also influences the amount and type of food available for wildlife. Possibly the most important role of fire in the longleaf ecosystem is its effects on vegetative composition and structure. Fire prunes back and/or reduces the number of woody stems, while enhancing diversity of herbaceous vegetation (Russell et al. 1999).

Fire and herpetofauna

Frequent fire maintains habitat structure, native fire adapted vegetation, and the invertebrate prey base that support the longleaf herpetofaunal community (Engstrom 1993; Engstrom et al. 2001; Mushinsky 1985). Although some studies have been conducted on direct effects of fire on herpetofaunas (Means & Campbell 1981), additional research is needed (Greenberg 2000; Russell et al. 1999; Schurbon & Fauth 2003).

Amphibian and reptile species respond differently to fire, as is exhibited in a few case studies of direct effects of burns. Lyon et al. (1978) suggested that smaller and less mobile vertebrates are more likely to remain in an area in response to fire and incur higher mortality compared to more mobile species. Amphibians, especially, exhibit poor dispersal capabilities, and may be more likely to experience high mortality in fire (Russell et al. 1999). Komarek (1969), however, suggested that amphibians and reptiles native to fire-maintained ecosystems have adaptive measures that allow them to minimize the rates of fire-induced mortality. Species from different geographic regions have diverse reactions in response to fire. For example, Western Fence Lizards (*Sceloporus occidentalis*) survive fires by seeking shelter under surface objects while Box Turtles (*Terrapene carolina*) and Eastern Mud Turtles (*Kinosternon subrubrum*) survive fires by burrowing into soil (Russell et al. 1999). In Australia, Friend (1993) reported a high incidence of amphibian species burrowing to avoid fire mortality.

Direct effects of fire on the longleaf herpetofaunal community are not well studied. There are observations in this ecosystem of many reptile and amphibian species

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using Gopher Tortoise (*Gopherus polyphemus*) burrows as cover from fire (Means & Campbell 1981). Other larger species of reptiles (e.g. lizards and snakes) may avoid fire mortality by simply dispersing from the burned areas (Means & Campbell 1981; Russell et al. 1999).

Indirect effects of fire on the longleaf herpetofaunal community may be mediated by alteration of vegetation. For example, long-term suppression of fire allows hardwood invasion, in turn reducing the habitat structure preferred by the Six-lined Racerunner (*Cnemidophorus sexlineatus*), Florida Sand Skink, Mole Skink (*Eumeces egregius*), Crowned Snake (*Tantilla relicta*), and Gopher Tortoise (Auffenberg & Franz 1982; Campbell & Christman 1982b; Greenberg et al. 1994; Means & Campbell 1981).

Thinning

Thinning methods avoid the risks presented by fire (loss of control and smoke) and may aid in diminishing hardwood tree species. Thinning results in a decrease in tree density, adds slash (logging debris) to forest floors and increases soil moisture and temperature (Smith et al. 1997). Thinning also reduces fire hazards by reducing fuel loads (when woody biomass is removed), and is beneficial for longleaf pine when it reduces competing trees (Litt et al. 2001b). Thinning may create openings in the canopy that allow light to reach the ground. In longleaf pine forests, this increase in light may facilitate establishment of seedlings (McGuire et al. 2001). Thinning gives longleaf more space, sunlight, and nutrients. In turn, these conditions maintain high species richness of the local flora and fauna.

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Although thinning can be conducted in a manner that mimics some of the results of natural fire, it can be harmful if not executed properly. In many thinning operations, heavy machinery is brought to a site to cut and clear trees. This machinery not only disturbs the soil and ground vegetation, it can cause erosion and/or compaction of the soil. This practice can disturb habitats, destroy food sources, and create boundaries that are barriers for small organisms. While thinning can be beneficial to the longleaf pine, it must be conducted in a responsible manner and may be more suitable for longleaf management if used in conjunction with other management treatments, most importantly, fire. Fire but not thinning, removes accumulated pine needles and ground cover litter.

Thinning and herpetofauna

Thinning may have variable effects depending on the herpetofaunal species in question. Campbell and Christman (1982b) suggested that clearcutting mimics wildfire in creating similar habitat features utilized by reptiles, especially in Florida sand pine scrub. In Oregon, Cole et al. (1997) studied effects of logging and broadcast slash burns on six species of amphibians. No logging effect was detected for Rough-skin Newts (*Taricha granulosa*), Dunn's Salamanders (*Plethodon dunni*), or Red-legged Frogs (*Rana aurora*), although Pacific Giant Salamanders (*Dicamptodon tenebrosus*) and Ensatinas (*Ensatina eschscholtzii*) incurred negative effects, and Western Red-back Salamanders (*Plethodon vehiculum*) responded positively to logging treatment. In Florida pine flatwoods communities, Enge and Marion (1986) found mean amphibian richness did not differ between minimum-(low impact, live vegetation remaining, minimal soil exposure and compaction) and maximum-treatment (heavy impact, removal of live vegetation and

soil compaction) clearcut sites. However, reptile species richness was lower in maximum treatment clear-cut sites than in minimum treatment sites. The study also indicated that clearcutting reduced amphibian abundance tenfold by affecting reproductive success. In Florida sand-pine scrub, Anderson and Tiebout (1993) suggest that piles of woody debris and the consequent reduction of the availability of bare ground may account for the reduced numbers of *C. sexlineatus* and *Sceloporus woodi* on recently burned sites. Greenberg et al. (1994) did not find the same results in their study of reptiles on sand-pine sites. Instead, they found no correlation in species composition between sites with high woody debris and sites with minimal levels of woody debris. However, they did find a trend towards lower numbers of *Tantilla relicta*, *C. sexlineatus* and *S. woodi* on burned plots.

Herbicide

Herbicide use is a topic of much debate. Brockway and Outcalt (2000) suggest that herbicides may be a useful management tool when used in conjunction with prescribed burns because this combination of activities can accelerate ecosystem restoration following periods of fire suppression. In the current study, application was limited to targeted individual plants (hardwood stems) and was specifically designed to eliminate contact with water sources.

A review of recent studies suggests both negative and positive effects of herbicide on vegetation. Some differences are likely due to different effects of specific herbicides and on the method of application. Herbicides are typically incorporated as management practices in order to reduce unwanted plant species as well as change the plant species

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composition and structure. A review of herbicide studies conducted by Litt et al. (2001a) found in many plantation studies weed control was the reason for herbicide application. The review also indicated a use of herbicides on flatwood vegetation reduced plant species richness and herbaceous woody plant cover, while increasing grasses and decreasing woody plant species cover on sandhill sites. Litt et al. (2001a) indicated herbicides may have a negative effect on non-target plant species in public and private southern pine lands. McCormack (1994) suggested most herbicide applications were used to improve species composition as well as increase volume and decrease rotation length. McComb and Hurst's (1986) review found that depending on the type of herbicide used, applications increase grass, forb, vine and woody vegetation as well as reduce hardwood shrubs when compared to control plots. Results from a seven-year study indicated herbicide treatments reduced hardwood basal area by greater than 55% in comparison to untreated plots (Quicke et al. 1996). In studies involving Red-cockaded Woodpeckers (*Picoides borealis*), the use of herbicide did little to improve the habitat and only marginally reduced unwanted plant species (Brennan et al. 1995). Other studies of the effects of herbicides on ground-layer vegetation have found few clear, consistent results (Litt et al. 2001a). Most forms of herbicide reduce invasive hardwoods and understory vegetation. However, herbicides alone lack the ability to provide many of the services that fire provides (Brennan et al., 1998).

Herbicide and herpetofauna

Studies on the effects of herbicide on reptile and amphibian populations report different results (Allran & Karasov 2000; Diana et al. 2000; Howe et al. 1998; Mann &

Bidwell 1999; Moore et al. 1998). These differences may be due, in part, to differences among herbicides and method of application, as well as differences among herpetofauna species. Herbicide treatments can create a habitat in which tree and/or shrub canopy cover is eliminated allowing for direct sunlight to penetrate to the forest floor, in turn creating an undesirable habitat for moisture-sensitive amphibians, as well as creating a desirable habitat for many reptiles (McComb & Hurst 1986). Negative effects may be offset by the increase in available invertebrate prey (McComb & Hurst 1986) and increased nest sites (Eubanks et al. 2003).

In Oklahoma, Jones et al. (2000) found herpetofaunal captures were lowest in herbicide-treated plots in comparison to herbicide with fire and untreated plots. An herbicide treatment with the addition of fire produced herpetofaunal numbers higher than the herbicide-alone treatments, but lower than that of the control. Species richness was not significantly different among treatments or control plots. Amphibians were least common on pastures that were treated with herbicide and fire, and reptiles were least common on herbicide-only pastures (Jones et al. 2000). Lizards were found to be more abundant on untreated plots and snakes were found in higher numbers on pastures treated with herbicide and fire (Jones et al. 2000). Studies on the Oregon coast found no effect of herbicide on herpetofauna capture rates (Cole et al. 1997).

To date, herbicide use has not been fully evaluated for overall influence on reptiles and amphibians. Chemical treatments can provide habitat improvements that are beneficial to herpetofauna and many studies of herbicide treatments have observed little or no effect on many reptiles. However, other studies have shown detrimental effects on amphibian reproduction and growth when chemicals are present in water sources (Forest

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Service 1984; Hayes et al. 2003; Lautenschlager & Sullivan 2002). Future studies are needed in order to quantify the effects of specific herbicides used in land management activities that affect herpetofauna.

Summary

Due to habitat loss and fragmentation it is increasingly important for proper human management to be conducted on the remaining tracts of longleaf forest. This management must take into account the needs of sensitive species, such as amphibians, and mimic the dynamics of the natural disturbance regime. Current literature emphasizes the need for further research on amphibians and reptiles in regard to long-term effects of forest management treatments such as the use of thinning and herbicide as alternatives to fire, determination of distribution and habitat requirements of herpetofaunal elements, and studies with rigorous experimental design (i.e. treatment replication and untreated sites; (Pilliod et al. 2003; Russell et al. 1999). Brennan et al. (1998) suggest that, if used, herbicide should be applied in conjunction with other management practices in order to provide benefits that herbicide alone cannot contribute.

CHAPTER 1

EFFECTS OF LONGLEAF PINE MANAGEMENT PRACTICES ON THE

HERPETOFAUNA OF SOUTH ALABAMA

Keywords: herpetofauna, longleaf pine, species diversity

Introduction

Historical accounts estimate that the Longleaf Pine (*Pinus palustris*) ecosystem

once covered approximately 37 million hectares of the southeastern United States, with

30 million hectares considered longleaf dominant and 7.3 million hectares representing

longleaf in mixed species stands (Frost 1993). This once vast forested landscape ranged

from the southeastern portions of Virginia, along the east coast to Florida, and west into

Georgia, Alabama, Mississippi, Louisiana, and East Texas. This ecosystem historically

was regulated by frequent (1-10 years) fires (Platt et al. 1991). The natural disturbance of

lightening-induced fire influenced the structure and composition of this ecosystem. With

the onslaught of anthropogenic encroachment, this ecosystem was rapidly reduced from

is original range. Logging, grazing, turpentining, and fire suppression have led to a

decline of the original longleaf-dominated ecosystem to its present-day estimate of 1.2

million hectares (Frost 1993). What remains today of the longleaf forest is found

primarily in small patches of Georgia, Florida, Alabama, Mississippi, North Carolina,

South Carolina, and Texas. The suppression of fire and conversion of longleaf pine

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forest to off-site pine plantations, agricultural lands and other uses have contributed to a decline of both the native flora and fauna of the longleaf pine ecosystem.

Amphibians and reptiles are important components of longleaf ecosystems because of their unusual abundance and diversity (Guyer & Bailey 1993). These animals evolved under the influence of fire, a feature that maintained open forested stands with appropriate nutritive and thermoregulatory resources to allow reproduction and to fill other social needs, i.e. territoriality (Litt 1999). However, decades of fire suppression and habitat alteration have changed the forest structure, thereby reducing habitat quality for several members of the herpetofauna. As a result, seven species of reptiles and amphibians within the geographic range of longleaf pine are listed as declining or threatened by the U.S. Fish and Wildlife Service (Brennan et al. 1998).

To better understand the impacts of current longleaf pine management techniques on this faunal group, rigorous research must be conducted. Many studies have evaluated mammal and bird populations in longleaf pine forests (Brennan et al. 1995; Engstrom 1993; Kantola & Humphrey 1990; Provencher et al. 2002; Rutledge & Conner 2002; Tucker et al. 2003), yet herpetofauna have been greatly understudied (Enge & Marion 1986; Greenberg et al. 1994; Litt 1999; Litt et al. 2001b). Of primary importance is study of the effects of stand thinning and the application of herbicides, two methods that are thought to mimic the beneficial effects of natural fire regimes in reducing fuel loads and restoring ecosystem function, and that might be used in place of fire where the application of prescribed burning is not possible. In this study we evaluate the effects these management tools have on the herpetofauna of the southeastern Gulf Coastal Plains. In particular, we determine whether species richness, evenness, and/or

composition are altered in the short term on sites affected by these tools. Additionally, we use species accumulation curves fit with the Michaelis-Menten equation to determine the likely asymptotic species richness and species accumulation rate on sites managed with these tools for the long term.

Methods

Study design

This study was conducted at the Solon Dixon Forestry Education Center (Dixon Center), a 2,165 ha tract of pinelands located approximately 29 km southwest of Andalusia, Alabama (Fig. 1). This site is situated in the Gulf Coastal Plains of south-central Alabama and includes habitats ranging from dry, sandy ridges containing Longleaf Pine and Turkey Oaks *(Quercus laevis)* to swamp wetlands containing Bald Cypress *(Taxodium distichum)* and Tupelo Gum *(Nyssa aquatica)*. The Dixon Center is managed by the Auburn University School of Forestry and Wildlife Sciences and is used for educational purposes, along with timber production, recreational activities, and research in association with the Longleaf Alliance and Auburn University (Hildreth 1999).

Five treatments; prescribed burning, forest thinning, thinning followed by prescribed fire, application of herbicide followed by prescribed fire, and untreated sites (no burning, thinning, or application of herbicide), were implemented on the Dixon Center (Fig. 1). Three replicates of each treatment were established. Each site was comprised of a 12.25 ha core area surrounded by a 20 m wide buffer creating a total site area of 15.2 ha.

Prescribed fires for the fire-only sites were completed during April and May of 2002. Fire application was done through manual use of drip torches. Thinning was conducted during late March and early April of 2002 (Appendix A). The regime applied to six sites involved a thinning-from-below improvement cut (Smith et al. 1997). The target trees for thinning on each site were hardwoods with a residual basal area of approximately 5.6 m^2 per 0.4 ha. To obtain the desired residual basal area some Loblolly Pines (*Pinus taeda*) were removed from all sites but only after removal of hardwoods, Spruce Pine (*Pinus glabra*) and Slash Pine (*Pinus elliottii*). All Longleaf Pines were left whenever possible. In areas with low basal area, some hardwoods were left standing to maintain canopy cover (D. Pancake, personal communication). A 4.0-4.5% solution of Garlon4 herbicide was applied to three sites in Fall 2002, followed by prescribed fire in late April on each site (Appendix A). Garlon4 was selected for its ability to penetrate waxy leaved plants, and for its limited direct effect on grasses and soil (R. Johnson, personal communication). Approximately 208-227 L of the Garlon4 solution was applied per 0.4 ha. Solo backpack sprayers were used to apply the herbicide to avoid impact on midstory species like dogwoods. Spraying occurred on all woody brush from ground level to 2.1 m above ground in late September after grasses and annual forbs had made seed. This treatment was followed by burns in April and May of 2003. Precipitation data was collected daily from seven established rain gauges located on the Dixon Center. Average monthly precipitation was significantly greater in 2003 in comparison to 2002.

Trapping design

One drift fence array and one snake funnel trap array were established on each of the fifteen sites. Each drift fence array was constructed of three sheets of aluminum flashing (15.24 m long and 60.96 cm tall). The arrays were arranged so that the three fences radiated from a central 18.9 L bucket that was sunk flush with the ground. The fences were placed so that angles of approximately 120° were created from the central bucket. Fences were placed upright in the ground by digging an 8-15 cm deep trench within which one edge of the fence was placed leaving approximately 45 cm above ground. An additional 18.9 L bucket was placed at the distal end of each fence [Figure 2A; arrangement modified from Campbell and Christman (1982a)]. Bucket lids were used to cover the buckets during times when sampling was not conducted. During sampling periods each lid was raised to allow entry of small organisms while also providing shade. Sponges were placed in buckets to avoid desiccation of trapped amphibians. Holes were placed in the bottom of each bucket to allow water drainage.

Funnel traps were constructed from a 50.8 cm x 91.4 cm piece of aluminum window screening folded into a rectangle with funnels located on each end. Funnel traps were placed on both sides of the middle of each fence for a total of 6 funnel traps for each pitfall array. Funnels were secured flush against the fence and ground by aluminum stakes. During non-sampling periods clips were used to close the funnels. A total of 45 drift fences with 60 pitfall and 90 funnel traps were established in June 2002.

Fifteen snake box traps were constructed. Each box was 1.2 x 1.2 x 0.46 m (Rudolph et al. 1999). A funnel was located in the middle of each of four sides of the trap to allow animals to enter (Figure 2B). Fences with dimensions of 61 cm x 9.1 m and

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constructed with hardware cloth (0.64 x 0.64 cm mesh) radiated from each of four sides of the snake box trap and served as drift fences. A box trap was established in an area adjacent to each drift fences in March of 2003.

Sampling

Traps were checked daily or every other day depending on capture rates. Date, site, trap type (bucket or funnel), location (cardinal direction) of capture, species, snout-vent-length (SVL in mm), and mass (in grams, excluding snakes) were recorded for each individual. Snakes were given individual marks by clipping ventral scutes. After data collection all specimens were released approximately four m from the trap (same side of fence for funnel traps; random direction for pitfall traps).

Time-constrained searches, a process that samples species that may not be captured in pitfall arrays (Campbell & Christman 1982a), were used to sample additional members of the herpetofauna. To avoid bias (Bury & Corn 1988), all time-constrained searches were performed by a single individual (AER). For each site, a fifteen-minute walking survey was conducted and repeated weekly. Date, site, species, and distance from observer were recorded for each individual encountered. Survey time, beginning approximately at 0700 hr, was rotated among sites to control for differing diurnal activities of the herpetofauna.

Herpetofaunal community analysis

Per-diem capture rates for 2002 and 2003 were calculated in order to standardize data between years due to unequal sampling periods. Pitfall trap rates were calculated by dividing the total number of individuals captured per site by the number of days that traps remained open. Per-diem observation rates from time-constrained data were calculated

from the total number of individuals observed on each site divided by the total number of days sampled. Time-constrained observation rates were then added to pitfall trap rates to give a total capture rate for each site.

Species diversity was evaluated based on two variables, species richness and evenness. Species richness was the count of the number of species present on each site. Evenness was derived from the Shannon index (Magurran 1988).

Capture rates and evenness were tested with a two-way [taxon (amphibians vs. reptiles) and treatment] repeated measures (year) Analysis of Variance (ANOVA). The tests were performed to document differences in capture rates and evenness among treatments and between taxa and years, as well as interactions among these variables. A two-way ANOVA was used to test for differences in species richness among treatments and taxa. Due to unequal sampling periods and low species richness, standardized rates were not calculated for species richness. This, in turn, prevented comparisons between years.

Species composition was analyzed on the basis of a presumed historical relationship of some species with the longleaf pine ecosystem. Guyer and Bailey (1993) established a list of resident and specialist amphibians and reptiles associated with longleaf pine. Residents were classified as species that have viable populations within longleaf pine, yet their distributional limits are not set by the longleaf ecosystem. A specialist was defined as a species that exhibits viable populations within longleaf pine and that have 80% or greater of its range overlapped by longleaf pine. A final category, not specifically named by Guyer and Bailey (1993), contained species with geographic ranges that overlap that of longleaf pines, but are not dependent on longleaf ecosystems

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for viable populations and, therefore, do not have the majority of their range in longleaf pine (generalist). Due to low numbers of specialist species recorded in this study, residents and specialists were pooled into a group; all others were considered generalist. A maximum likelihood test (SAS) was used to determine if treatments differed in the number of longleaf specialist/resident versus generalist species (Appendix B).

Jaccard's index was used to analyze similarities in species composition among sites. Jaccard's index equals 1 in the case of complete similarity (all species in common) and 0 when sites are completely dissimilar [no species in common; (Magurran 1988)]. Jaccard's indices were calculated for each treatment comparison (i.e. burn vs. thin, thin vs. untreated sites, etc.). Each combination of comparisons (three sites for each treatment) was then averaged for an overall treatment-by-treatment similarity index. For example, all three burned sites were compared to all three thinned sites for a total of nine combinations. Values for each combination were averaged in order to attain a treatment-by-treatment index. Indices from 2002 were compared to indices from 2003, thus documenting trends in similarity over time.

Species accumulation curves were constructed for each site. Samples were divided into thirty-day increments, for nine sampling periods. New species were tallied for each sampling period, adding to the cumulative number of species captured. Plots of accumulated species were fit using the Michaelis-Menten equation (Raaijmakers 1987). This equation was used to project the point at which additional sampling yielded a 0.05 or less increase in species richness. Both the number of days required to reach this point and species richness at this point were used to evaluate how well the herpetofauna had been sampled during the study period.

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Burn, thin, and thin-and-burn treatments were analyzed separately and compared to untreated sites because this design was fully crossed and all treatments were implemented in 2002. These four treatments will be referred to as the core treatments. The herbicide-and-burn treatment was not crossed with other treatments and was not implemented until 2003. Therefore, herbicide-and-burn sites were compared with untreated sites in separate analyses. An alpha level of 0.05 was used for all statistical tests. Alpha values ranging from 0.06 to 0.09 were considered to be marginally significant.

Results

A total of 1018 and 6416 individuals were recorded for 2002 and 2003, respectively. For both years, amphibians comprised the majority of captures. In 2002, amphibians were 82.4% (81.6% Anura and 0.8% Caudata) and reptiles (all Squamata) were 17.6% of all captures. Total captures in 2003 were comprised of 92.1% amphibians (90.9% Anura and 1.2% Caudata) and 7.9% reptiles (7.6% Squamata and 0.1% Testudines).

In 2002, 15 species of amphibians were captured, 12 Anura, 3 Caudata, and 15 species of reptiles (all Squamata). Fifty-one species were captured in 2003, including 14 Anura, 9 Caudata, 23 Squamata, and 5 Testudines. *Bufo terrestris* comprised 34%, and *Gastrophryne carolinensis* comprised 39% of total captures in 2002. *Bufo terrestris*, *Gastrophryne carolinensis*, and a third anuran, *Scaphiopus holbrookii*, dominated the capture rates in 2003 with 38%, 29%, and 17% of observations, respectively.

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Capture rates differed significantly between taxa, with reptiles having consistently higher daily capture rates than amphibians ($F = 29.78$; p<0.0001; df = 1; Fig.3). No significant difference was found between years ($F = 0.35$; p = 0.56; df = 1) or among core treatments ($F = 2.47$; p = 0.1; df = 3) and no combination of interactions between years, core treatments, or taxa was significant. For herbicide-and-burn and untreated site comparisons a significant difference between taxa was found, with reptile daily capture rates being higher than amphibian capture rates ($F = 9.67$; p = 0.01; df = 1). There was no significant difference between years ($F = 1.46$; p = 0.26; df = 1) or treatments ($F = 0.002$; p = 0.96; df = 1). However, a marginally significant interaction was found between time and taxon ($F = 4.48$; p = 0.07; df = 1) because reptile capture rate decreased in 2003 while amphibians increased (Fig 3). No other interaction term approached significance.

Species richness did not differ among core treatments for 2002 or 2003 ($F = 0.18$; p = 0.91; df = 3, $F = 0.46$; p = 0.71; df = 3, respectively) or between taxa ($F = 1.32$; p = 0.27; df = 1, $F = 2.09$; p = 0.17; df = 1, respectively). The interaction of these two variables did not differ for either year ($F = 0.47$; p = 0.70; df = 3, $F = 0.59$; p = 0.62; df = 3; 2002, 2003 respectively; Fig.4). No difference in species richness between herbicide-and-burn and untreated sites or between amphibians and reptiles was observed for 2002 or 2003 ($F = 0$; p = 1; df = 1, $F = 0.36$; p = 0.56; df = 1 respectively, for treatments; $F = 3.20$; p = 0.11; df = 1, $F = 0.81$; p = 0.39; df = 1, respectively for taxa; Fig. 4). There was no taxon*treatment interaction in 2002 or 2003 ($F = 0.8$, p = 0.40; df = 1, $F = 2.27$; p = 0.17; df = 1, respectively).

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Evenness differed significantly between years ($F = 31.30$; $p < 0.001$; df $= 1$), with greater evenness in 2002 than in 2003, and between year and taxon ($F = 72.3$; $p < 0.0001$, df $= 1$) with greater amphibian evenness than reptiles in 2002 and an inverse relationship in 2003 (Fig. 5). No significant difference occurred between taxa ($F = 3.05$; $p = 0.1$; df $= 1$), among treatments ($F = 0.78$; $p = 0.42$; df $= 3$) or among interactions ($F = 0.15$; $p = 0.52$; df $= 3$ for taxon*treatment, $F = 1$; $p = 0.42$; df $= 3$ for year*treatment, $F = 0.17$; $p = 0.91$; df $= 3$ for year*taxa*treatment). When comparing herbicide-and-burn and untreated sites, evenness differed significantly between years ($F = 6.80$; $p = 0.03$; df $= 1$), with higher evenness in 2002 than 2003, and between year and taxon ($F = 42.92$; $p = 0.0002$; df $= 1$) with more amphibians than reptiles in 2002 and an inverse relationship in 2003 (Fig. 5). No significant difference occurred between taxa ($F = 0.0018$; $p = 0.97$; df $= 1$), among treatments ($F = 0.09$; $p = 0.78$; df $= 1$), or for interaction effects ($F = 0.04$; $p = 0.85$; df $= 1$ for taxon*treatment, $F = 0.56$; $p = 0.47$; df $= 1$ for year*treatment, $F = 0.003$, $p = 0.96$, df $= 1$ for year*taxon*treatment).

In examining species composition for the core treatments, a maximum likelihood test indicated that the proportion of specialist taxa did not differ among treatments ($\chi^2 = 0.21$; $p = 0.98$; df $= 3$) or with the interaction of treatment and years ($\chi^2 = 1.69$; $p = 0.64$; df $= 3$). The proportion of specialists and generalist on herbicide-and-burn and untreated sites was not different between treatments ($\chi^2 = 1.73$; $p = 0.19$; df $= 1$) or for the interaction of treatment and years ($\chi^2 = 0.16$; $p = 0.69$; df $= 1$; Table 1). Jaccard's indexes indicated that, in 2002, core treatments shared 0.36 to 0.45 of their species. In 2003, treatments became more similar to each other, with indices ranging from 0.42 to 0.51, a trend that was significant ($F = 16.48$; $p = 0.0002$; df $= 1$). No significant differences

were found among treatments ($F = 0.79$; $p = 0.56$; df = 5), or in the interaction of treatment and year ($F = 1.77$; $p = 0.14$; df = 5). Herbicide and burn and untreated site comparisons were significantly different between years ($F = 7.17$; $p = 0.03$; df = 1) with a mean Jaccard index of 0.42 in 2002 and 0.49 in 2003.

Sampling effectiveness and species estimators

Species accumulation curves estimated a mean of 49 species on burn-only sites, 56 species on thin-treated sites, 43 species on thin-and-burn treated sites, 33 species on herbicide-and-burn treated sites, and 31 species on untreated sites (Fig 6A). Mean estimated species richness did not differ among core treatments ($F = 1.19$; $p = 0.37$; df = 3) and was about half the observed number of species (Fig. 6A). Similarly, herbicide-and-burn and untreated sites did not differ between estimated species means ($F = 0.08$; $p = 0.79$; df = 1). There was a significant difference in the amount of time required to reach the estimated number of species for core treatments ($F = 5.18$; $p = 0.03$; df = 3), with thin treatments requiring approximately twice as many sampling periods as burn and thin-and-burn, and three and a half times as many sampling periods as untreated treatments. There was no difference in sampling periods required to estimate species richness of herbicide-and-burn and untreated sites ($F = 0.60$; $p = 0.48$; df = 1; Fig 6B).

Discussion

The three treatments (thinning, burning, and application of herbicide) are all known to create a more open aspect to stands of longleaf pine and this feature of ecosystem structure is the principle goal of restoration efforts in longleaf habitats

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(Provencher et al. 2001). We expected this change in vegetation structure to increase the densities of specialist species of the longleaf pine community while decreasing the densities of generalist taxa. Therefore, we expected to observe an altered herpetofaunal composition due to treatment effects. Overall, our results indicate that, if such treatment effects are occurring, they happen over a time interval longer than the current study period. We found no short-term (two-year) treatment effects on capture rates, diversity (richness and evenness), proportional composition of specialist and generalist taxa, or similarity of species composition. However, when projected over the long-term, we found a significant treatment effect for the rate at which an asymptote for the number of species is reached on core treatments. Species accumulation curves reach an asymptote on untreated sites faster than burn and thin-and-burn sites, all of which reach asymptotes faster than on thinned sites. Because this order of core treatment effects also reflects the order for estimated number of species on each treatment (lowest species richness on untreated sites, intermediate on burn and thin-and-burn sites, highest on thinned sites), the significant treatment effect appears to indicate both a change in the shape of the Michaelis-Menten curves (steepest in untreated sites, shallowest in thinned sites) and the target number of species. Our data suggest that herbicide-and-burn had no effect on expected species richness or the rate at which that richness was reached relative to untreated sites. We conclude from this that the herpetofauna is altered by the core treatments but that the time frame over which such community-level effects will occur is longer (6-21 years) than the time period of our study. This time frame is consistent with patterns of amphibian monitoring projects in other regions of the southeastern United States (Pechmann et al. 2001). Population data for the three most common species

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suggest that the treatments increased densities of specialist taxa and decreased densities of generalist taxa (Chapter 2), but we have no way to project whether this ultimately will translate into our expected alteration in species diversity by increasing the overall contribution of specialist taxa to diversity and decreasing the contribution of generalist taxa. Our data indicate that the effect of fire is the same whether performed alone or in conjunction with stand thinning and that thinning alone results in projected patterns of species richness that are in the same direction as burning but accumulate more slowly and reach a larger cumulative species richness.

Variation between years and between taxa played a much stronger role in explaining patterns of occurrence of amphibians and reptiles than did treatments. Rainfall increased significantly in 2003 relative to 2002. We argue that this feature caused the significant differences in capture rates and evenness observed in this study. In 2003, reptile capture rates declined and amphibian capture rates increased. Associated with these changes, the evenness component of species diversity was greater for reptiles and lesser for amphibians in 2003 than in 2002. We suggest that rainfall increases activity for amphibians but that this increase is greater for some species than for others, creating assemblages of species that are dominated by common species. Reptile activity does not appear to be altered by varying levels of precipitation, and we suggest the increase in evenness in 2003 may be due to the implementation of snake box traps during this trapping season.

Our study is one of the few replicated designs examining effects of management practices on the herpetofauna of longleaf forests (but see Litt et al. 2001b). Although the study sites were selected to minimize variation among plots, such variation still existed.

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Despite this variation, the management tools used in this study should have resulted in consistent differences among treatments. However, the effects of a single application of these treatments might be short lived. For example, a fire frequency of every two years may be required to create habitat appropriate for Bachman's sparrow (Tucker 2002). Similarly, gopher tortoises will abandon longleaf pine habitats in as little as 5-7 years if vegetation is not maintained with an open canopy (Aresco & Guyer 1999). Our study was performed only for a single burn event and this occurred on plots that had not been burned for 3 to 5 years previously. Similarly, herbicide-and-burn was applied only a single time. We suggest treatments may need replication before a strong effect on herpetofaunal communities are seen. For both amphibians and reptiles, the species composition became more similar across all treatments in 2003 relative to 2002. We suggest that this pattern indicates that treatments were likely to have resulted in effects that altered the assemblage of amphibians and reptiles but that these short-term effects were weak following a single treatment. Difference may not be obvious and may not be maintained until a series of treatments have been applied.

Conclusion

This study indicates that there are no immediate community-level effects of burning, thinning or herbicide treatments on amphibians or reptiles of longleaf pine forests. However, the possibilities of long-term effects were suggested by our data. These effects indicate that thinning, and/or burning are likely to yield habitats in which herpetofaunal diversity will require long periods of time to assess and where species richness may be increased. However, the effects of these two management tools are

likely to require frequent re-application to maintain. We found no evidence that application of herbicide along with fire would replicate the effect of fire alone. Thinning does provide similar community-level effects on the herpetofauna as burning does and so thinning appears to be an appropriate substitute for fire, where burning cannot be performed.

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Appendix A. Description of the 15 study plots.

Site	Block	Hectares	Last Burn	Treatment	Thinning date	Burn date	Herbicide
1	3	-	-	Burn	N/A	15-May-02	N/A
2	1	14.2	1997	Thin and Burn	27-Mar-02	12-Apr-02	N/A
3	1	34	1999	Thin	27-Mar-02	N/A	N/A
4	1			Herbicide and Burn	N/A	May	23-Sep-02
5	1	32	1999	Untreated	N/A	N/A	N/A
6	1			Burn	N/A	17-Apr-02	N/A
7	2	20	1998/99	Herbicide and Burn	N/A	13-May-03	28-Sep-02
8	2			Untreated	N/A	N/A	N/A
9	2	30.4	1999	Thin	31-Mar-02	N/A	N/A
10	2			Thin and Burn	31-Mar-02	22-May-02	N/A
11	2	10.5	1999	Burn	N/A	21-May-02	N/A
12	3	52.6	1999	Herbicide and Burn	N/A	May	30-Sep-02
13	3			Untreated	N/A	N/A	N/A
14	3			Thin and Burn	4-Apr-02	1-May-02	N/A
15	3	10.1	1998	Thin	4-Apr-02	N/A	N/A

Appendix B. List of species known or suspected to occur within the longleaf pine forest of the Solon Dixon Center. S = specialist; R = resident; G = generalist; * = observed at Solon Dixon Center during this study.

TAXON	RESIDENT/SPECIALIST GENERALIST	CAPTURED
Amphibia		
Ambystomatidae		
Ambystoma cingulatum	S	
A. maculatum	G	*
A. opacum	G	*
A. talpoideum	R	*
A. tigrinum	R	*
Amphiumidae		
Amphiuma means	G	
A. pholeter	G	
A. tridactylum	G	
Plethodontidae		
D. apalachicolae	G	
D. auriculatus	G	
D. conanti	G	
E. chamberlaini	R	
E. longicauda	G	
Eurycea cirrigera	G	*
Hemidactylium scutatum	N	
P. glutinosus "complex"	R	*
Psedotriton montaunus	N	
P. ruber	N	*
Proteidae		
N. "lodingi"	N	
Slamandridae		
N. viridescens	R	*
Sirenidae		
Siren intermedia	N	
S. lacertina	N	
Anura		
Bufonidae		
B. quercicus	S	
B. terrestris	S	*
B. woodhouseii	R	
Hylidae		
Acris crepitans	R	*
Acris grylluls	R	*
Hyla andersonii	R	
H. avivoca	R	
H. chrysoscelis	R	*
H. cinerea	R	*
H. femoralis	R	*
H. gratiosa	S	*
H. squirella	S	*
P. crucifer	R	*

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Species		
P. nigrita	S	
P. ornata	S	*
P. feriarum	R	
Microhylidae		
Gastrophryne carolinensis	R	*
Ranidae		
R. capito	S	
R. catesbeiana	G	*
R. clamitans	G	*
R. grylio	G	
R. heckscheri	G	
R. okaloosae	G	
R. sphenocephala	R	*
Pelobatidae		
Scaphiophus holbrooki	R	*
Crocodilia		
Alligatoridae		
Alligator mississippiensis	G	
Squamata		
Anguidae		
Ophisaurus attenuatus	R	
O. mimicus	R	
O. ventralis	G	*
Phrynosomatidae		
Sceloporus undulatus	R	*
Polychrotidae		
Anolis carolinensis	R	*
Scincidae		
E. egregius	R	
E. fasciatus	R	*
E. inexpectatus	R	
E. laticeps	R	*
Scincella lateralis	R	*
Teiidae		
Cnemidophorus sexlineatus	R	*
Colubridae		
Carphophis amoenus	R	
Cemophora coccinea	R	*
Coluber constrictor	R	*
Diadophis punctatus	G	*
Drymarchon corais	R	
Elaphe guttata	R	*
E. obsoleta	R	*
Farancia abacura	G	
F. erytogramma	G	
Heterodon platyrhinos	R	*
H. simus	S	
Lampropeltis calligaster	R	
L. getula	R	
L. triangulum	R	*
Masticophis flagellum	R	*
N. erythrogaster	G	
N. fasciata	G	*
N. floridana	G	
N. sipedon	G	
N. taxispilota	G	

Opheodrys aestivus	G	
Pituophis melanoleucus	R	*
R. rigida	G	
R. septemvittata	G	
Rhadinaea flavilata	S	
Seminatrix pygaea	G	
Storeria dekayi	G	
S. occipitomaculata	G	*
Tantilla coronata	R	*
T. sauritus	G	
T. sirtalis	G	*
Virginia striatula	R	
V. valeriae	R	
Elaphidae		
Micrurus fulvius	R	
Viperidae		
Agkistrodon contortrix	R	*
A. piscivorus	R	*
Crotalus adamanteus	S	*
C. horridus	R	*
Sistrurus miliarius	R	*
Testudines		
Chelydridae		
Chelydra serpentina	G	
Macroclemys temminckii	G	
Emydidae		
Chrysemys picta	G	
Deirochelys reticularia	G	*
G. emsti	G	
P. concinna	G	
P. floridana	G	
Terrapene carolina	R	*
Trachemys scripta	G	*
Kinosternidae		
K. subrubrum	G	*
S. minor	G	
S. odoratus	G	
Testudinidae		
Gopherus polyphemus	S	*
Trionychidae		
Apolone ferox	G	
A. mutica	G	
A. spinifera	G	

Table 1. Contingency table for number of specialist and generalist species on core treatments, herbicide-and-burn treatments and untreated sites.

A

2002		
Treatment	Specialist	Generalist
Untreated	17	3
Burn	14	2
Thin	17	1
ThinBurn	19	2
2003		
Treatment	Specialist	Generalist
Untreated	25	6
Burn	28	7
Thin	28	9
ThinBurn	29	5

B

2002		
Treatment	Specialist	Generalist
Untreated	16	2
HerbicideBurn	14	0
2003		
Treatment	Specialist	Generalist
Untreated	25	6
HerbicideBurn	26	3

Figure Legend

Figure 1. Map of Solon Dixon Center showing location of treatment sites. See appendix
B for details of treatments

Figure 2. Trapping designs used for collecting samples. A) Pitfall trap array; shaded
circles = bucket traps; x = funnel traps; line indicates aluminum flashing. B)
Snake box trap

Figure 3. Effects of treatment and time on mean daily capture rate of amphibians A) and
reptiles B) Dark bars indicate 2002 capture rates. Open bars indicate 2003
capture rates. Vertical bars represent + 1 mean standard error.

Figure 4. A) Mean species richness for amphibians. B) Mean species richness for
reptiles. Dark bars indicate mean species richness for 2002. Open bars indicate
mean species richness for 2003. Vertical bars represent + 1 mean standard error.

Figure 5. Effects of treatment and time on mean evenness of amphibians A) and reptiles
B). Dark bars indicate 2002 capture rates. Open bars indicate 2003 capture rates.
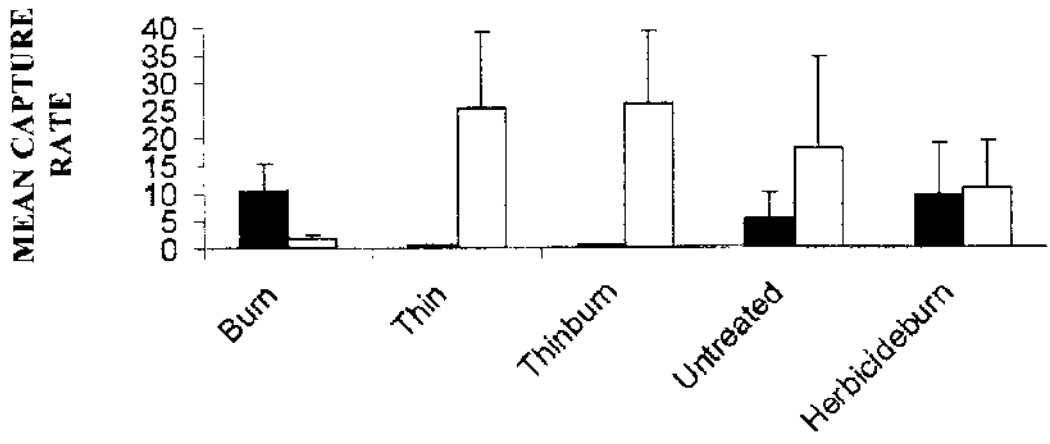Vertical bars represent + 1 mean standard error.

Figure 6. Estimated species and sampling periods. A) Open bars indicate mean observed
species. Dark bars indicate Michaelis-Menten estimated number of species. B)
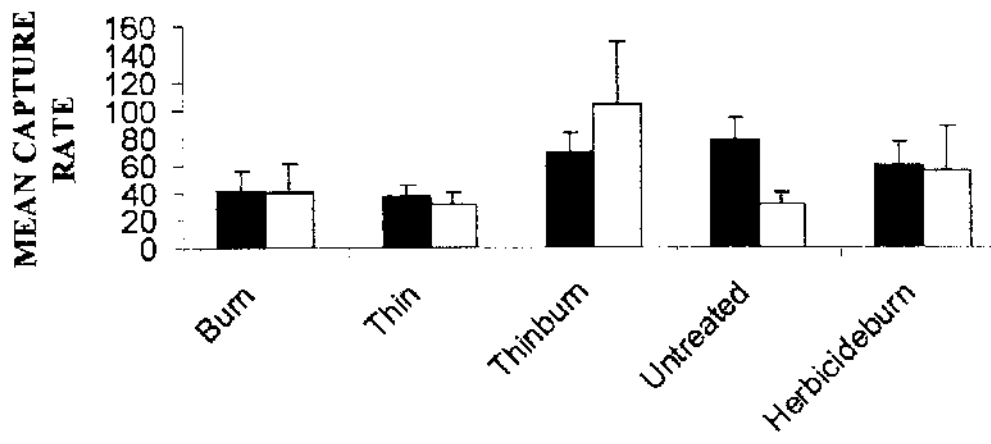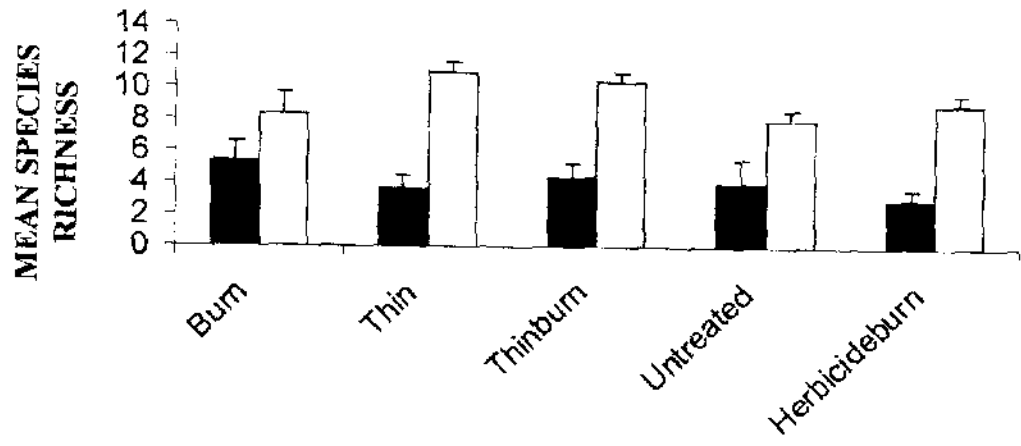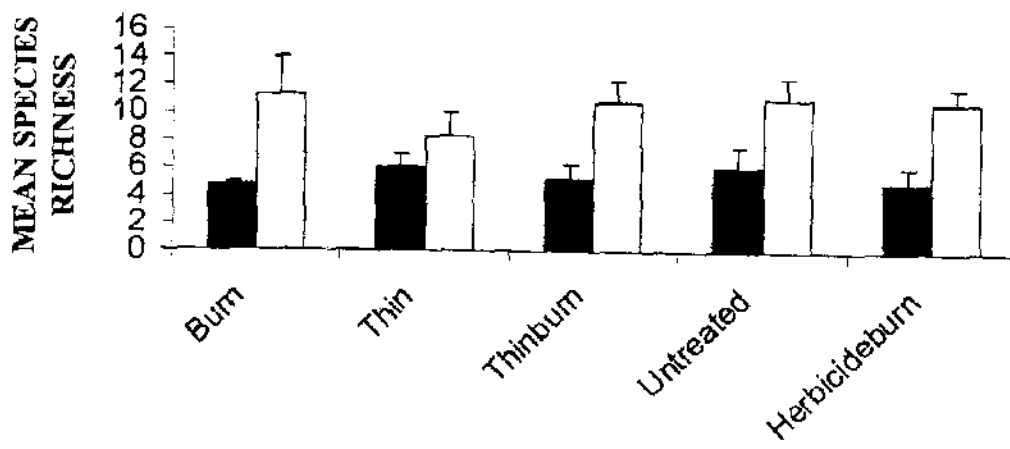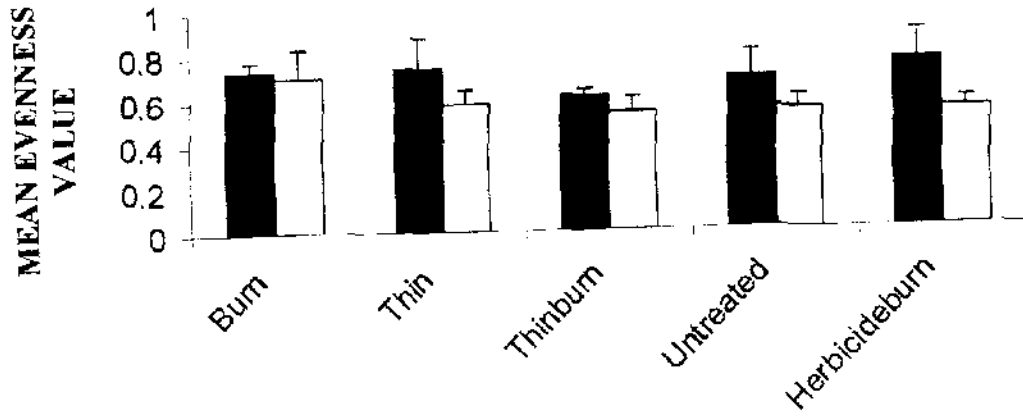mean estimated sampling period needed to reach 95% of estimated species.

Burn

Untreated

Herbicide & Burn

Thin

Thin & Burn

Center Rd.

Hwy 29

A

15.24 m

⬤ Pitfall

X Funnel trap

B

A

B

41

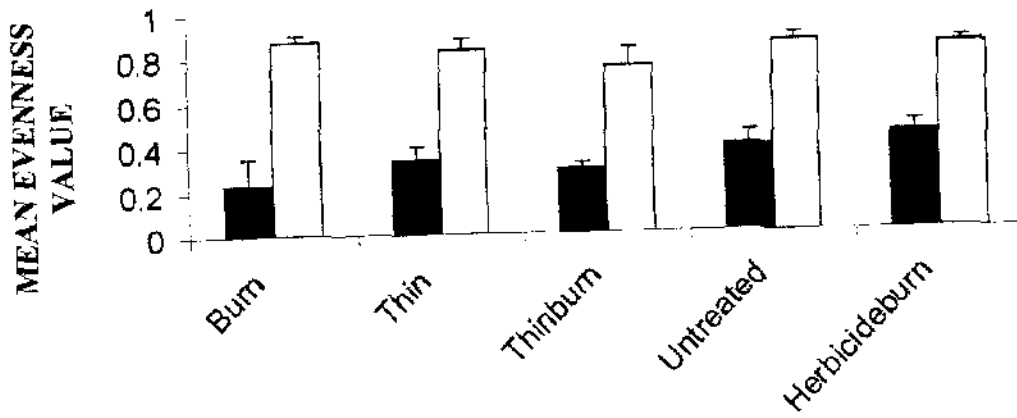**A**

B

42

A

B

43

A

B

44
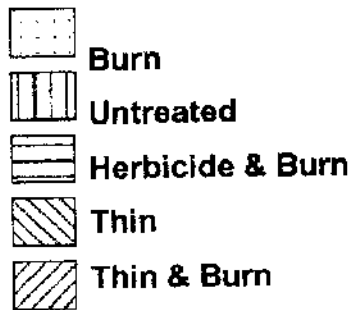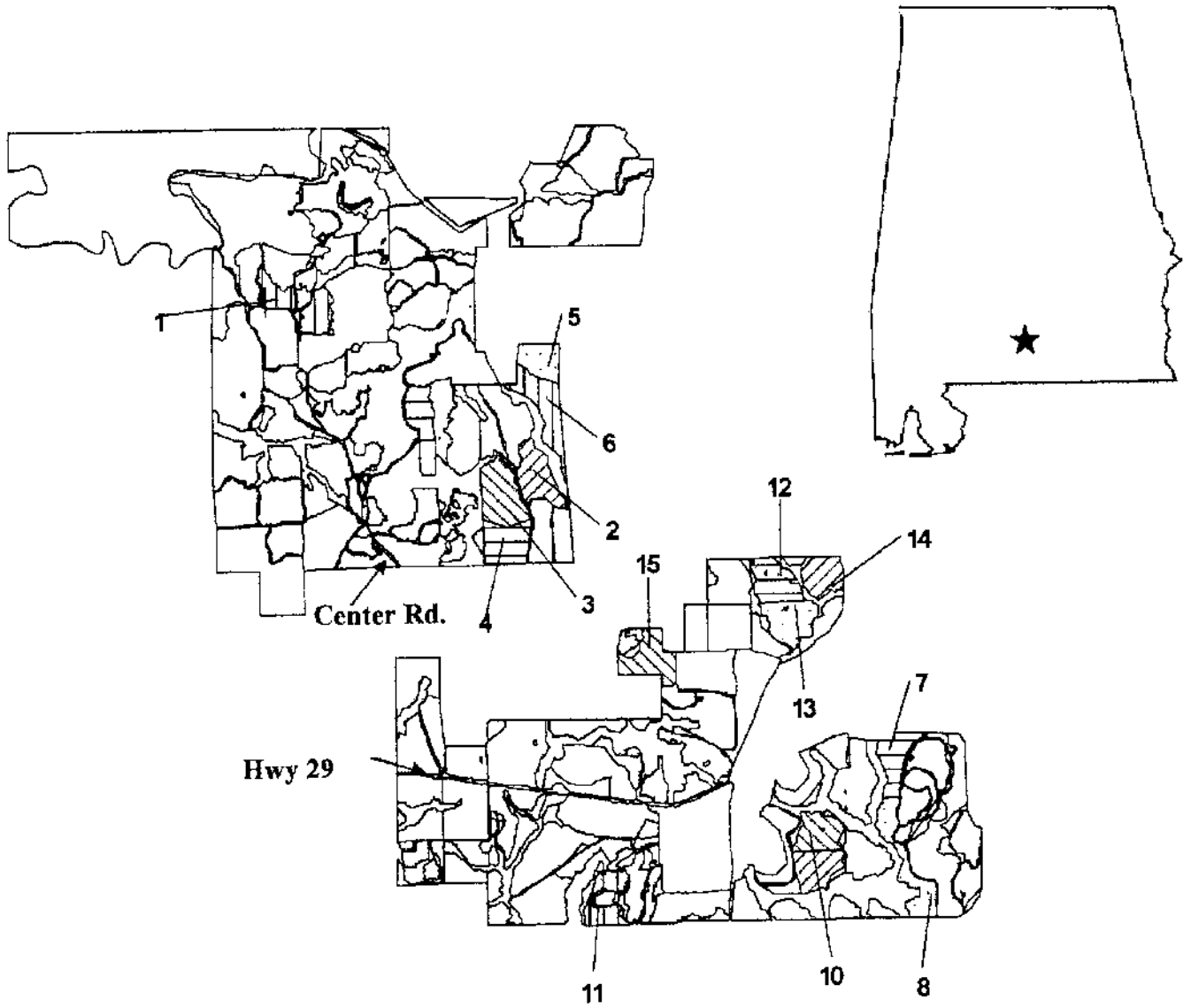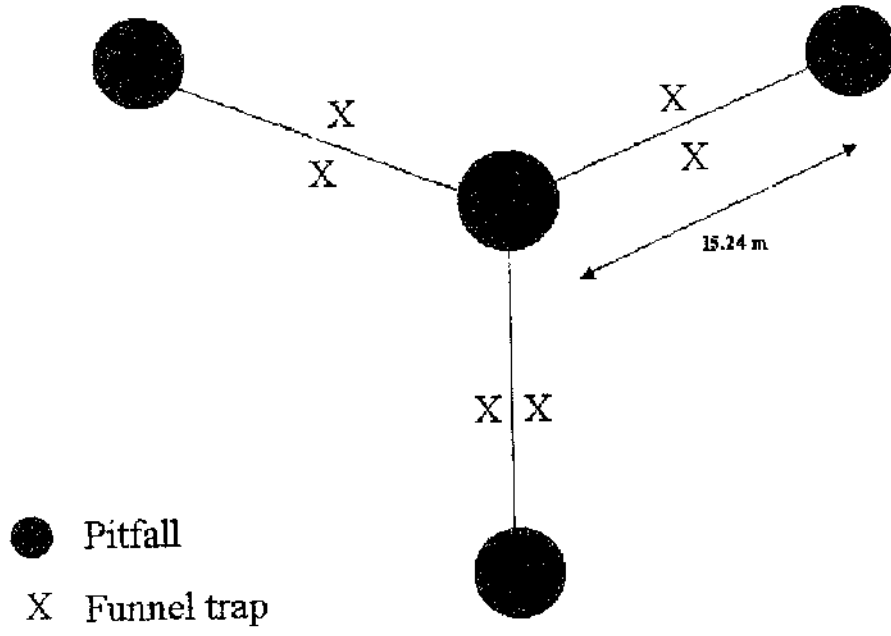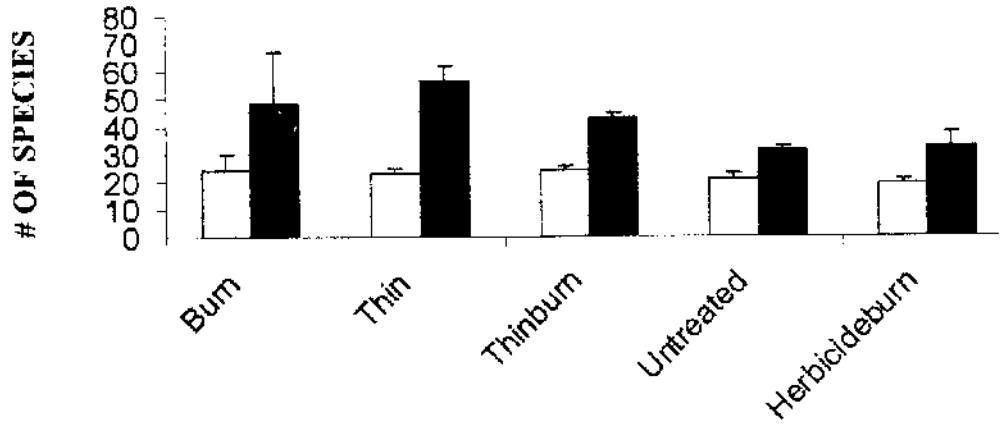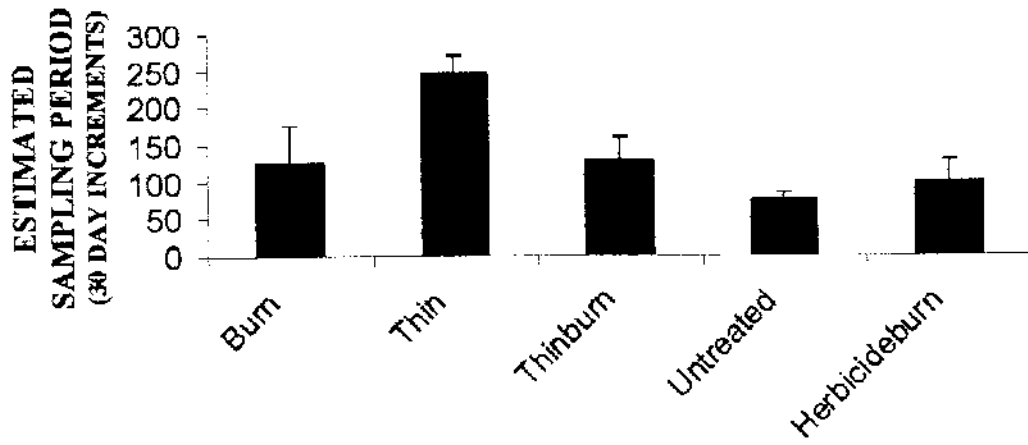
CHAPTER 2

EFFECTS OF LONGLEAF PINE MANAGEMENT PRACTICES ON *BUFO*

TERRESTRIS, GASTROPHRYNE CAROLINENSIS, AND *SCINCELLA LATERALIS* OF

SOUTH ALABAMA

Keywords: herpetofauna, longleaf pine, species diversity

Introduction

Many herpetofaunal species within the Longleaf Pine (*Pinus palustris*) ecosystem

have their life histories entwined with the natural processes that occur in that ecosystem

whereas others appear to be adapted to other habitats but can survive in longleaf. Both

sets of species serve as important parts of the food web and contribute to the unusual

species richness of this component of the fauna (Guyer & Bailey 1993). However, the

two types of species differ in their values to conservation and restoration efforts

associated with the longleaf ecosystem. There is increasing concern for species that are

limited to longleaf pine ecosystems (Noss et al. 1995). With the onslaught of

anthropogenic encroachment, the longleaf pine, and its herpetofauna have suffered losses

due to deforestation, forest fragmentation, and prevention of natural regenerative fires in

remaining forested areas (Brennan et al. 1998; Frost 1993). This process frequently has

altered the ancestral landscape in ways that have impoverished species adapted to the

longleaf pine ecosystem and expanded populations of species adapted to other habitats.

Thus, specialist species, those closely associated with longleaf pine ecosystems, may be displaced when, during fire exclusion, hardwood shrubs and trees increase in dominance at the expense of longleaf pine and associated plant species. Eventually these specialists may be replaced with more generalized (generalist) species (Guyer & Bailey 1993). This process has been noted in other fire-maintained ecosystems where some herpetofaunal components develop improved demographics on burned sites and others become impoverished on such sites (Cavitt 2000; Clawson et al. 1997; Enge & Marion 1986; Greenberg et al. 1994; Litt et al. 2001b; McLeod & Gates 1998; Mushinsky 1985; Perison et al. 1997).

In order to maintain the original fauna of longleaf ecosystems it is important to know how management practices designed to restore longleaf habitats affect the ancestral biota adapted to this forest type and the potential competitors that may have invaded or increased as the result of habitat degradation. We focus on three common herpetofauna species and address the effects of prescribed burning and two management tools used to mimic aspects of the beneficial effects of burning, stand thinning, and application of herbicides. These effects are examined for populations of *Bufo terrestris*, a species that, while present in many habitats within the longleaf ecosystem, is most characteristic of the sandy soils inhabited by longleaf specialist taxa, and for *Gastrophryne carolinensis*, and *Scincella lateralis*, two species that are characteristic of hardwood forests and increase in numbers in longleaf habitat when fire is withheld. We chose these species because they are abundant enough to allow us to gather sufficient data to determine treatment effects and because they are representative of groups that we expected to respond positively (specialists; increased abundance, biomass, and body condition) and negatively

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(generalists; decreased abundance, biomass, and body condition) to the restoration of key features of the longleaf pine ecosystem.

Methods

As previously detailed in Rall and Guyer (Chapter 1), three replicates of burn, thin, thin-and-burn, and herbicide-and-burn treatments were established on the Solon Dixon Forestry Education Center, a field station located approximately 29 km southwest of Andalusia, Alabama. Three untreated (no burning, thinning or application of herbicide) sites were also established for comparative analysis. Each site was comprised of a 12.25 ha core area surrounded by a 20 m wide buffer, creating a total site area of 15.2 ha. Sites were selected so that initial conditions were as similar as possible and treatments were assigned at random. One pitfall array (drift fence with four 18.9 L buckets and 6 funnel traps; see Chapter 1) was established on each of the fifteen sites. Herpetofaunal sampling was conducted from 23 May through 4 August 2002, and from 9 April through 24 December 2003. Date, site, trap site, trap location (cardinal direction) of capture, species, snout-vent-length (SVL in mm), and mass (in g) were recorded for each individual captured in a trap. Additional sampling was conducted through time-constrained searches of each site during daylight hours. A single individual (AER) conducted all walking surveys in order to avoid observer bias. Site of capture and date were recorded for all individuals of the three target species.

Species analysis

Rates of capture for pitfall traps were calculated by dividing the total number of individuals captured per site by the number of days that traps remained open. Per diem observation rates from time-constrained data were calculated from the total number of individuals observed on each site divided by the total number of days sampled. Time-constrained per diem observation rates were then added to per diem pitfall trap rates to give a total capture rate for each site. Biomass was calculated as the sum of the masses of all individuals within a species divided by the number of trap days. This method accounted for differing trap rates between years. Body condition for each species was determined from residuals of log_{10}SVL (snout-vent length) plotted against log_{10}mass. Positive residuals indicate increased body condition and negative residuals indicate decreased body condition.

Capture rates and biomasses were analyzed with a one-way (treatment) repeated measures (year) Analysis of Variance (ANOVA). Due to unequal capture rates, comparison of body condition among treatments was analyzed with a one-way ANOVA calculated separately for each year. Most *S. lateralis* were observed during time-constrained searches and so no data were recorded for mass or SVL. Therefore, biomass and body condition comparisons for *S. lateralis* were not made.

Burn, thin, thin-and-burn, and untreated treatments will be referred to as core treatments and were analyzed separately from herbicide-and-burn treatments. The core treatments were analyzed separately because they represent a fully crossed design and were implemented in the same year (2002). The herbicide-and-burn treatments were not performed until 2003 and this treatment was not crossed with the other three. Therefore,

herbicide-and-burn treatments were compared only to the untreated sites. Significance was determined among treatments and between years, as was the significance of the interaction between these variables. An alpha of 0.05 was used to determine significance. Alpha values ranging from 0.06-0.09 were considered to be marginally significant.

Results

For capture rates of *B. terrestris* and *G. carolinensis* no significant difference was found among treatments ($F = 0.59$, $p = 0.64$, $df = 3$; $F = 0.93$, $p = 0.47$; $df = 3$, respectively), between years ($F = 3.53$, $p = 0.1$, $df = 1$; $F = 1.03$, $p = 0.33$, $df = 1$; respectively) or in the interaction of treatment and year ($F = 0.92$, $p = 0.47$, $df = 3$; $F = 1.05$, $p = 0.42$, $df = 3$; respectively; Fig. 1). Capture rates between herbicide-and-burn and untreated treatments for *B. terrestris* and *G. carolinensis* did not differ significantly among treatments ($F = 0.04$, $p = 0.84$, $df = 1$; $F = 3.67$, $p = 0.13$, $df = 1$; respectively), between years ($F = 0.23$, $p = 0.66$, $df = 1$; $F = 4.73$, $p = 0.1$, $df = 1$; respectively) or in the interaction of treatment and year ($F = 0.24$, $p = 0.65$, $df = 1$; $F = 4.61$, $p = 0.10$, $df = 1$; respectively; Fig. 1). Capture rates for *S. lateralis* differed significantly among core treatments ($F = 4.33$, $p = 0.04$, $df = 3$), with higher capture rates on untreated sites than other sites (Fig. 1). The effects of year and the interaction between treatment and year were not significant ($F = 1.30$, $p = 0.29$, $df = 1$; $F = 0.86$, $p = 0.50$, $df = 3$; respectively). Comparison of herbicide-and-burn and untreated sites yielded no significant difference ($F = 2.24$, $p = 0.21$, $df = 1$), and did not differ between years ($F = 0.21$, $p = 0.67$, $df = 1$).

The interaction of treatment and year was not significant ($F = 2.27$, p = 0.21, df = 1; Fig 1).

For core treatments, biomass of *B. terrestris* was marginally significantly different between years ($F = 3.71$, p = 0.09, df = 1), with a higher biomass in 2003 (Fig. 2). No significant differences were found among treatments ($F = 0.51$, p = 0.69, df = 3), or in the interaction of treatment and year ($F = 0.14$, p = 0.93, df = 3). Herbicide-and-burn and untreated treatments differed significantly between years ($F = 7.76$, p = 0.05, df = 1) with a higher biomass for *B. terrestris* in 2003 (Fig. 2). No difference was found among treatments ($F = 1.24$, p = 0.33, df = 1) or in the interaction of treatment and year ($F = 2.75$, p = 0.17, df = 1). Biomass of *G. carolinensis* did not differ among core treatments ($F = 0.93$, p = 0.47, df = 3) or between years ($F = 0.87$, p = 0.38, df = 1; Fig. 2). No significant interaction between treatment and year was found ($F = 4.11$, p = 0.75, df = 3). Biomass of *G. carolinensis* did not differ between herbicide-and-burn and untreated treatments ($F = 2.56$, p = 0.18, df = 1) or years ($F = 3.97$, p = 0.12, df = 1; Fig. 2). The interaction of treatment and year was not significant ($F = 4.13$, p = 0.11, df = 1).

Body condition of *B. terrestris* did not differ among core treatments in 2002 ($F = 1.29$; p = 0.28, df = 1), but did differ in 2003 ($F = 3.13$, p = 0.03, df = 3) when body condition was greatest on thin sites followed by burn sites. Herbicide-and-burn vs. untreated treatment comparisons did not differ in 2002 ($F = 0.29$, p = 0.59, df = 1), but did differ significantly in 2003 ($F = 5.14$, p = 0.02, df = 1; Fig 3) when body condition was greater on herbicide-and-burn sites. Similarly, core treatment comparisons of *G. carolinensis* body condition were not significantly different for 2002 ($F = 0.29$, p = 0.83, df = 3), but were significantly different for 2003 ($F = 9.75$, p < 0.0001, df = 3) when

body condition was greatest on untreated sites followed by thin sites. Herbicide-and-burn vs. untreated treatment comparisons did not differ in body condition during 2002 ($F = 0.67$, $p = 0.42$, df = 1) but did differ in 2003 ($F = 4.23$, $p < 0.0001$, df = 1; Fig. 3) when body condition was greater on untreated than herbicide-and-burn treatments.

Discussion

As predicted, *S. lateralis*, a species typical of hardwood areas, decreased in capture rate on plots that were opened through fire and thinning. We found no difference in the effect of these two management tools on the magnitude of decline in capture rate, and no additive or multiplicative effect of a combination of them. So, both thinning and fire seem equally viable as a mechanism to reduce increases of this and perhaps other potential competitors to the longleaf specialist herpetofauna. Because the effect occurred rapidly, the reduced encounter rates may have been the result of direct mortality caused by the treatment or from rapid migration of individuals from the treatment plots to nearby, less open habitat. Fire is known to cause direct mortality in glass lizards (Means & Campbell 1981), but because it is unlikely that thinning would result in direct mortality to *S. lateralis* and because the effects of thinning and burning were equivalent, we argue that immigration from the treatment plots is the most likely explanation for the reduced encounter rates. However, a year later we found no such effect for herbicide application followed by fire, a finding that does not fit the pattern of the other treatments. We have no explanation for why fire did not have the same proposed effect on the herbicide-and-burn sites.

Contrary to our expectation, capture rates of *B. terrestris* did not increase on treatment plots and capture rates of *G. carolinensis* did not decrease on these plots. This suggests that the treatment effects were not strong enough to affect the demography of these two amphibian species or that there was insufficient time following treatments for abundances to be altered through reproduction and/or migration. Information from body condition (see below) suggests that the latter is more likely.

Biomass comparisons for *B. terrestris* were marginally significantly different between years. Precipitation may play a key role in this result. Greater rainfall in 2003 may have contributed to greater amphibian activity and indirectly may have increased prey resources through increased vegetation growth. Such an increase in food availability may have resulted in greater amphibian biomasses. Herpetofaunal community analysis showed this trend in taxon and year effects (see Chapter 1). We suggest greater amounts of rainfall positively influences amphibians and has little effect on reptiles in terms of captures.

In 2003, body condition of *B. terrestris* was greater on thin sites and burn sites compared with untreated sites. Similarly herbicide-and-burn sites had toads with greater body condition than untreated sites. Only the thin-and-burn sites failed to demonstrate increased body condition relative to untreated sites, but these data trended in that direction. Stand thinning and application of herbicide-and-burn reduced hardwood vegetation, creating an open understory and likely increasing ground temperatures. Burn sites created an open understory canopy and a loss of forest debris, in turn creating an environment in which new understory vegetation could grow. This new vegetation may have provided a food source for herbaceous insects, in turn providing prey items for

toads. Untreated sites, due to a lack of disturbance, may not have created an environment in which new vegetation could grow, leaving not only a closed understory canopy, but also a lack of new vegetation and possibly fewer prey insects.

We suggest that the overall pattern indicates improved body condition of toads on sites that are opened due to thinning, burning, or application of herbicides. Like all members of the genus, *B. terrestris* has thick and bumpy skin that retards water loss, characteristics that may enhance the success of this species in the open canopy of longleaf forests. Environmental factors on treatment sites may allow *B. terrestris* to inhabit areas that are not best suited for species normally associated with nearby hardwood-dominated forests.

Gastrophryne carolinensis had greater body condition on untreated sites in comparison to core treatment and herbicide-and-burn sites. Retention of forest debris on untreated sites is likely to have been the key variable leading to improved body condition of *G. carolinensis*. Due to its smooth moist skin, this species may be more susceptible to the effects of increased heat and decreased moisture than is *B. terrestris* (Behler 1998). Forest debris may be important for *G. carolinensis* to regulate body moisture as well as provide food and shelter for termites and ants upon which *G. carolinensis* feeds (Behler 1998). Thin sites were second to untreated sites in producing improved body condition in *G. carolinensis*. Because these sites were not burned, they retained woody debris that was not present on core treatments that received fire. The difference in body condition between thin and untreated sites may reflect pre-existing differences among these sites and/or reduction of debris during the thinning process. Similarly, body condition decreased on the herbicide-and-burn plots. This result may be related to the application

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of a fire in late spring of 2003 and not necessarily because of the direct effect of herbicides.

Gastrophryne carolinensis is a hardwood-adapted species found within the distributional range of longleaf pine but not considered a specialist of this habitat type (Behler, 1998). Management practices used to regenerate and maintain longleaf pine ecosystems are not of an overall benefit to this species. If longleaf pine forests are to be managed for both the native flora and fauna, then it appears that burning alone creates an environment that reduces levels of generalist taxa like *G. carolinensis*. The effects of burning on generalist taxa were not immediately duplicated by thinning or application of herbicides because the latter two treatments do not eliminate accumulated woody debris directly. However, accumulation rates of woody debris in areas opened by thinning or application of herbicides eventually are likely to be reduced to levels that would allow reduction of such dead vegetation via decomposition.

Our results demonstrate the value of recognizing and emphasizing goals for specialist species as the targets for habitat management and restoration. In addition, responses of generalist species to a land management treatment should be of less consequence for conservation concerns. Reptiles and amphibians of the longleaf ecosystem have a strong evolutionary tie in association with this habitat. Due to the extensive alteration of longleaf pine ecosystems from old-growth conditions (Simberloff 1993), the encompassing herpetofauna is of high conservation concern (Guyer & Bailey 1993). Restoration of conditions conducive to retaining these species in the landscape should be of high priority. Our approach differs from others who have based management recommendations for longleaf ecosystems on a goal of increased total

species richness (deMaynadier & Hunter 1995; Schurbon & Fauth 2003). Such studies often allow beneficial effects on common generalist species to mask deleterious effects on unique or endemic species (deMaynadier & Hunter 1995). For example, Schurbon and Fauth (2003) suggest decreasing prescribed fire frequency from every 2-3 years to 3-7 years, an inter-fire interval that is longer than that of the historical fire regime for many areas within the longleaf pine ecosystem (Platt et al. 1991). Our results suggest that this management decision that is likely to increase species richness at the expense of depressing longleaf endemic species for generalist, hardwood forest ones. By focusing management decisions on improving habitat for specialist taxa, land managers should be able to insure that the taxa that are most threatened are retained in the landscape.

Conclusion

Our data supports the use of longleaf pine management treatments as desirable techniques in creating habitats favorable to longleaf pine herpetofaunal specialists. Future studies may consider focusing on sampling individual species of concern, increasing treatment replicates, and sampling for a longer time period. We recommend caution when characterizing benefits and drawbacks of a large-scale management technique on a diverse herpetofaunal community.

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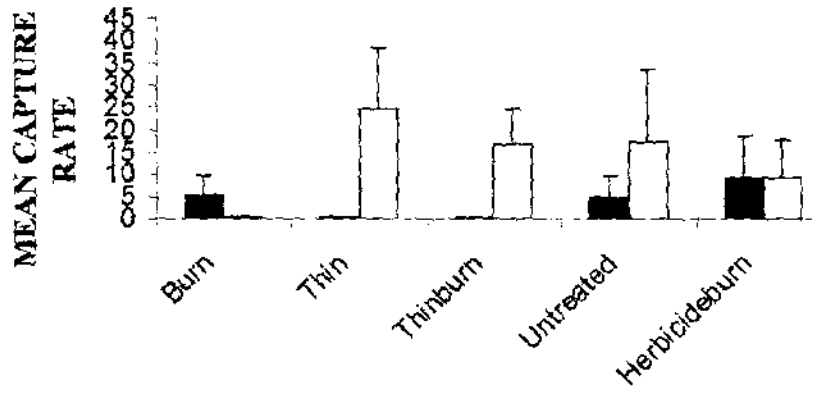
Figure Legend

Figure 1. Mean capture rate for three common species on treatment and untreated sites. Dark bars, data for 2002; open bars, data for 2003. Vertical bars indicate + 1 mean standard error. A) *Bufo terrestris*; B) *Gastrophryne carolinensis*; C) *Scincella lateralis*.
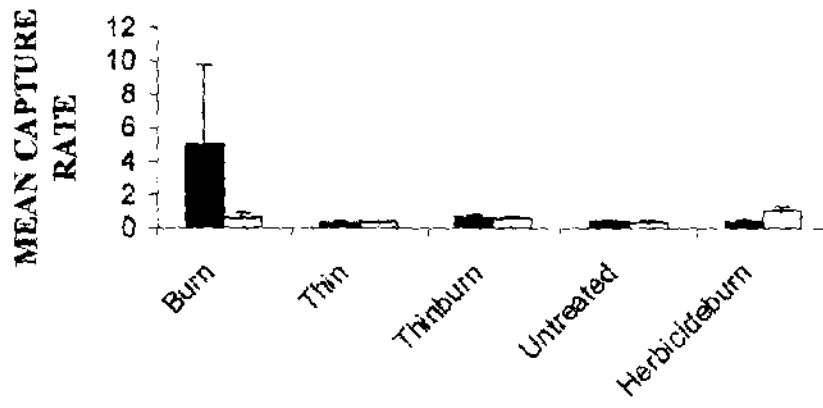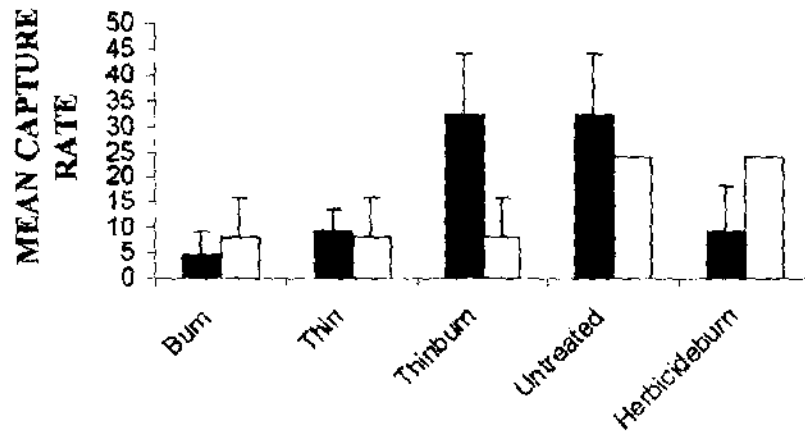
Figure 2. Mean biomass of two common amphibians on treatment and untreated sites. Dark bars, data for 2002; open bars, data for 2003. Vertical bars indicate + 1 mean standard error. A) *Bufo terrestris*; B) *Gastrophryne carolinensis*.
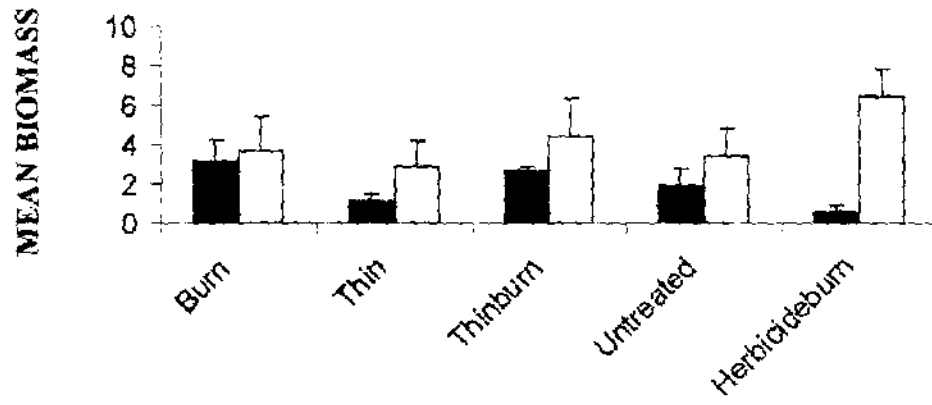
Figure 3. Mean body condition of two common amphibian species on treatment and untreated sites. Dark bars, data for 2002; open bars, data for 2003. Vertical bars indicate + 1 mean standard error. A) *Bufo terrestris*; B) *Gastrophryne carolinensis*.
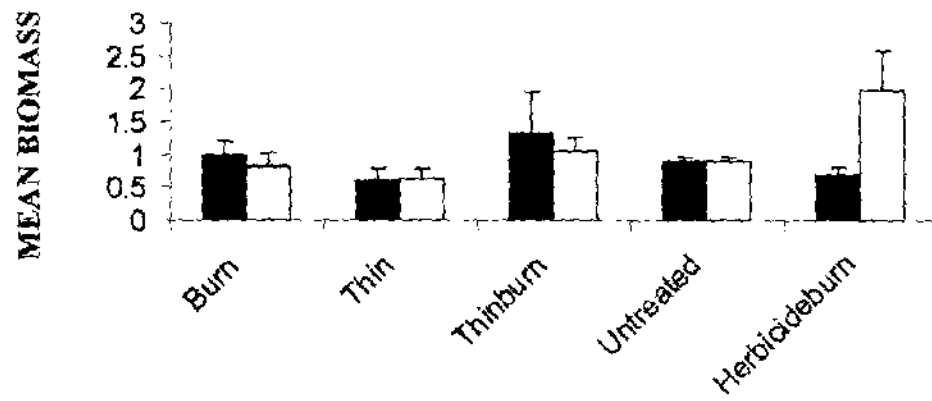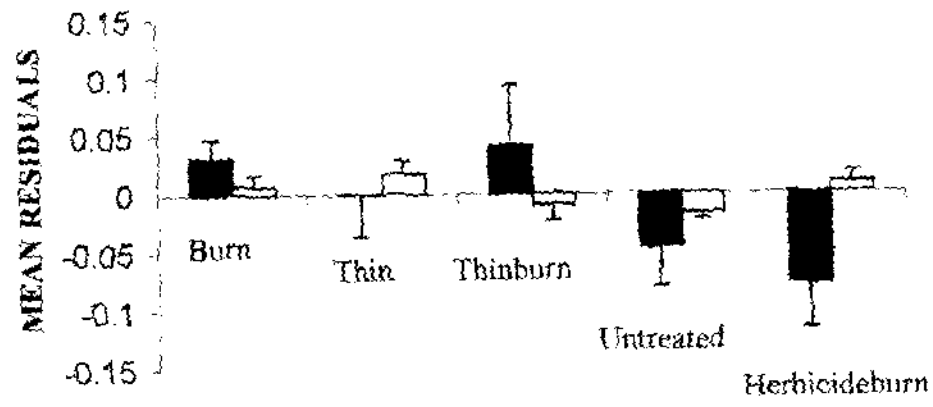
A

B

C

59

A

B

A

B