Herpetofaunal Responses to Restoration Treatments of Longleaf Pine Sandhills in Florida

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Abstract

The hypothesis that habitat restoration will provide for community reestablishment and the creation of habitat heterogeneity was examined with regards to the herpetofauna of longleaf pine sandhills in northwest Florida. The herpetofaunal response to restoration was examined in fire-suppressed, hardwood-dominated areas treated with (1) spring fire; (2) felling or girdling; or (3) a granular form of the herbicide hexazinone. No-treatment controls were also included. Felling or girdling and herbicide plots were burned for fuel reduction two dormant seasons after initial treatment application. Additionally, data were collected in frequently burned reference sandhills to establish the target condition or restoration goal. Vegetation variables and herpetofaunal capture rates were compared among control and treatment areas. Two similarity indices were utilized to compare treatments and controls with reference sites, to examine restoration success. Restoration treatment effects were observed through reduced hardwood densities. Litter composition varied among control and treatment plots, with leaf litter being highest in areas lacking recent fire. Capture rates of some herpetofaunal species varied significantly among treatment plots. In 1997 similarity indices showed that spring-burned and felling or girdling plots were more similar to the reference sandhills than the other plots. Treated plots were not significantly different from controls in 1998, a year of a severe drought.

Key words: **longleaf pine sandhills, restoration, herpetofaunal community, reference condition, fire, herbicide, mechanical reduction, similarity, habitat heterogeneity.**

Introduction

Restoration of degraded ecosystems is based partly on the assumption that rebuilding the habitat will result in the renewal of ecosystem processes and rehabilitation of faunal communities. Degraded areas under restoration will become increasingly similar to high integrity systems, representative of the reference condition. Whether habitat restoration is sufficient for community reestablishment has been discussed as the "Field of Dreams" hypothesis: "If you build it, they will come" (Palmer et al. 1997:295). Two aspects of this hypothesis are discussed below: the return of vegetation structure and creation of habitat heterogeneity.

Habitat structure and diversity have been shown to be very important in maintaining a diverse herpetofaunal community (Pianka 1966, 1967; Lillywhite 1977; Campbell & Christman 1982; Mushinsky 1992). In a study of Florida sandhills and scrub, Campbell and Christman (1982) suggested that the herpetofaunal community was determined by physical and biotic factors, as opposed to ecosystem types, because many of the same species can be found in different habitats with similar structural characteristics. Consequently, changes in habitat structure would likely shift the competitive balance and alter community composition (Lillywhite 1977; Means & Campbell 1981; Humphrey et al. 1985; Mushinsky 1985, 1986, 1992). To illustrate, Mushinsky (1992) showed that changes in the physical habitat structure resulting from fire were favorable for some lizard species, yet unfavorable for others (see also Campbell & Christman 1982; Patterson 1984; Mushinsky 1985). For example, fire suppression increases litter abundance, which may be beneficial for herpetofaunal species dependent on chemoreception for prey detection, because these species can easily forage in and under litter (Mushinsky 1992). However, high amounts of litter may result in declines in other species and shifts in species dominance (Caughley 1985).

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Microhabitat heterogeneity may be the most important factor determining herpetofaunal species diversity (Enge & Marion 1986). Herpetofaunal species diversity is correlated to microhabitat heterogeneity (Lillywhite 1977; Mushinsky 1985; Guyer & Bailey 1993; Greenberg et al. 1994*b*). It has been shown that a habitat with high microhabitat diversity is better able to fulfill the needs of a diverse herpetofauna under varying conditions (Braithwaite 1987; Paulissen 1988; Mushinsky & Gibson 1991; Mushinsky 1992).

Restoration of the habitat structure does not guarantee that species composition will converge toward a reference condition. Many of the studies described above compare sites under different disturbance regimes, or a nontreated control versus a treated site without the benefit of appropriate site randomization or without a reference site to establish target conditions. Restoration of a degraded system requires knowledge of the historic, native state (National Research Council 1992; MacMahon & Jordan 1994; Palmer et al. 1997) to develop restoration targets and measure success. Reference sites must be representative of the site-specific presettlement conditions to minimize large regional variability (White & Walker 1997). However representative, relatively pristine sites may be hard to find.

Even after reference sites are found, measuring restoration success is difficult for at least two reasons. First, tracking ecological change following restoration requires a large amount of data to identify sensitive metrics and increase statistical power. Second, direct comparisons to the reference condition are not statistically feasible because reference sites cannot be replicated and randomized for inclusion in the experimental design. Despite these difficulties, one must still determine how similar species assemblages or groups of variables are between restored plots and the reference condition. Similarity indices provide an intuitive way to examine restoration success.

This study was initiated to examine the efficacy of several restoration techniques on the sandhill herpetofaunal community. The “Field of Dreams” hypothesis (Palmer et al. 1997) was examined. It assumes that the restoration of habitat structure will restore the composition of the herpetofaunal community (regardless of the source within or outside of the plot) and habitat heterogeneity (regardless of patch size or configuration). Further, we examined the similarity of restored areas to high-quality sandhills or reference sites.

Methods

This study was conducted at Eglin Air Force Base (EAFB) in the western Florida panhandle. The base, located in Santa Rosa, Okaloosa, and Walton counties, encompasses 185,600 ha. This location has approximately 275 frost-free days per year and an average annual temperature of 19.4°C (Chen & Gerber 1990). Rainfall peaks during the summer months and averages 158 cm per year (DoD–Air Force 1995). The base is frequently subjected to lightning, tornadoes, and tropical storms (NOAA 1994). Soils in these sandhill areas are mainly of the Lakeland Series, with infrequent patches of Troup soils (Rodgers & Provencher 1999).

Historically, much of EAFB was used for timbering or the turpentine industry (DoD–Air Force 1993), activities that have changed the ecosystems on the base. Seventy-eight percent of EAFB’s land base are *Pinus palustris*-dominated (longleaf pine) sandhills (FNAI 1994). Today, many of these forests have been fire-suppressed and extensively logged, and hardwoods have fully encroached into the midstory.

This study is one component of the Longleaf Pine Restoration Project (LPRP) conducted by The Nature Conservancy, Tall Timbers Research Station, and the University of Florida. The LPRP’s primary objective is to test the effects of hardwood removal techniques on soil chemistry, vegetation, arthropods, and birds in formerly fire-suppressed sandhills. Three hardwood reduction treatments were examined: (1) spring burning (B); (2) mechanical felling or girdling of hardwoods (F); and (3) herbicide application (H). In burn plots, prescribed fires were conducted in April, May, and early June to simulate the natural disturbance historically experienced in this system (Myers 1990). In the felling/girdling treatment, hardwoods were felled or girdled with chainsaws and the slash was not removed. A granular form of the herbicide hexazinone, ULW (E. I. du Pont de Nemours, Wilmington, DE) was broadcast from the ground at a rate of 1.68 kg active ingredient/ha, one-quarter of the recommended application rate (Wilkins et al. 1993). ULW, a soil-active herbicide commonly used in southeastern forestry, is carried to plant roots and absorbed after rain (Gonzalez 1985; Berish 1996). All three hardwood reduction treatments were applied in the spring and early summer of 1995. Felling/girdling and herbicide treatments also were burned for fuel reduction in March and April 1997. Plots where fire suppression was maintained were used as controls (C).

Study plots were selected if they met certain criteria (Rodgers & Provencher 1999), mainly if they contained high densities of large hardwoods. Treatments were assigned to 81-ha study plots based on a randomized complete block design (Steel & Torrie 1980). This study utilized four blocks, for a total of 16 restoration plots (4 [3 treatments + control] $\times$ 4 blocks), on the western side of the base. Examination of aerial photos from 1940, 1969, 1983, and 1993, did not reveal any signs of fire, although thinning has occurred at varying levels in each of these blocks (Provencher et al. 2001*c*).

Reference (R) sites were selected specifically to represent the restoration target or goal and are not treatments or controls because they cannot be replicated or randomized in time or space. These 81-ha plots were located near test and training ranges (established between the late 1960s and the early 1980s) and thus experienced fairly frequent fire due to military activity (Rodgers & Provencher 1999). This study used four reference sites.

One uppercase"T"-shaped trapping array was placed at the center of each plot, with the direction randomly determined. This shape was selected to maximize the trapping area, while minimizing the number of traps. Each array comprised eight pieces of drift fence (four per line of the "T") constructed with galvanized steel flashing, 7.6 m long and 50 cm high, and 16 pitfall traps (2 traps per fence), which were plastic 19-L (5 gallon) buckets. This totaled 160 segments of drift fence and 320 traps in 20 plots.

Trapping was conducted from 23 May through 13 August 1997 and again from 9 April until 4 August 1998. Generally, traps were checked every other day; left open for 12 days, then closed for 2 days. All captured reptiles and amphibians were identified to species, sexed, individually marked, and released.

This study contains several known biases. The lack of pretreatment data in this examination must be kept in mind, for factoring out differences that were originally present (e.g., due to topography and the proximity to water), which could be mistaken for treatment differences. (Nearby creeks and streams were fairly equally distributed among treatments.) Additionally, pitfall traps and drift fences are especially good for capturing small, surface-active reptiles and amphibians (Vogt & Hine 1982; Greenberg et al. 1994*a*; Enge 1997). However, this method will not likely capture arboreal frogs, large turtles, or large snakes; their biology (e.g., size, behavior) and ability to trespass the fences and escape from traps precluding their being captured (Gibbons & Semlitsch 1981; Vogt & Hine 1982; Dodd 1991; Franz et al. 1995). We were less interested in species able to move long distances (e.g., large snakes), because they are less likely to be affected by these treatments. In addition, the assumption of independent observations is violated by the animals' movement among the treatment areas. The timing of this survey also dictated the species captured due to seasonal activity patterns (Vogt & Hine 1982; Franz et al. 1995). However, these biases were assumed to be present in all areas, thus permitting comparisons among study plots (Bury & Corn 1987; Corn 1994).

Vegetation sampling was conducted on all study plots ($n = 20$) at the beginning and end of each trapping season (May and August 1997, April and August 1998) and averaged for each year. Line transects (Bonham 1989) were used to measure the percent coverage of various cover classes including: bare ground, needle litter, non-needle litter, woody litter (>3 cm and <3 cm diameter), burnt litter (such as partially consumed bark, fine litter), grasses, *Pteridium aquilinum* (bracken fern), *Licania michauxii* (gopher apple), forbs, woody plants, woody vines (e.g., *Smilax* spp.), *Serenoa repens* (saw-palmetto), and *Yucca flaccida* (yucca). These cover classes were examined based on their structural properties, response to fire, or hypothesized effect on the herpetofauna (e.g., on locomotion, thermoregulation).

A 5-m line transect was established for vegetation sampling, parallel to each piece of drift fence, aligning the midpoints of the transect and fence. Transect placement (i.e., the side of the fence and the distance from the fence [2–5 m]) was randomly determined. These line transects were sampled four times during the study. Eight line transects, one for each piece of fence, were measured for each plot. The line transect was subdivided into 500 1-cm segments, and the number of segments where each cover class was present at approximately 1.5 m or lower was recorded. Because several vegetative cover types could be present in any given centimeter, the sum of all values could exceed 500; and because the mere presence of a cover class was recorded for each centimeter, finely structured cover items (e.g., pine needles) may be overestimated.

Belt transects (Bonham 1989) were used to determine tree density. A 4 × 7.5–m belt transect was centered along the midline of each fence. All trees in this area (>1 m high) were recorded using the following type and diameter categories: *Quercus* spp. (oak) (<5, 5–15, or >15 cm dbh), *Pinus* spp. (pine) (<5, 5–15, or >15 cm dbh), and other species (nonoak/pine, such as *Ilex* spp., *Diospyros virginiana*) (<5 or 5–15 cm dbh). Overstory cover was determined at the ends of each fence using a concave spherical densiometer (Lemmon 1957).

To address the second portion of the "Field of Dreams" hypothesis, that of habitat heterogeneity, we used Shannon's index of evenness (Zar 1996:40), which allowed us to determine if cover types were evenly distributed across the habitat. A value close to 1 would indicate a high evenness value or a relatively equal representation of all cover types. Hereafter, we will consider this a measure of habitat heterogeneity (see also Pianka 1967). This may seem counterintuitive; however, from the perspective of herpetofaunal species, if more cover types are available to satisfy habitat requirements for more species, this would indicate higher heterogeneity. The objective of this index was not to measure optimum patch size for individual species, but the different kinds and amounts of substrates that would accommodate as many species as possible. These values were transformed using x-square to homogenize variances and examined using ANOVA.

Two similarity indices were used to make comparisons between each reference site and each restoration plot: a widely used index, proportional similarity (Brower et al. 1989), and a new index that incorporates variability, endpoint difference (Provencher et al. 1999). Similarity was calculated to indirectly compare treatments based on their resemblance to the reference condition, without direct tests against the reference sites, which are not part of the experimental design. Calculations of similarity values and determinations of the strongest contributors were essentially the numeric equivalent to constructing species rank curves for each treatment and measuring goodness of fit to the reference condition. Numeric values provide the added advantage of using statistics for comparisons.

Both indices were used to obtain similarity values for groups of variables including herpetofaunal species capture rates, litter cover groups (needle litter, non-needle litter, bare ground, burnt litter, woody litter), and vegetation species cover (bracken fern, gopher apple, yucca, saw-palmetto, grasses, forbs, woody plants, and woody vines).

Proportional similarity (PS) (Brower et al. 1989) was calculated as:

$$PS_{ij} = 1 - 0.5 \sum_{k=1}^{s} (|p_{ik} - p_{jk}|),$$

where p is the proportion of species k in treatment plot i and in reference site j. The proportions are based on values for capture rates for all herpetofaunal species captured or relative abundance for vegetation cover variables. The logarithm of the variables was taken to increase the relative contribution of rare species. This formula was calculated for every restoration plot (n = 16), paired with each reference site (n = 4), and averaged over all reference sites per restoration plot. Proportional similarity will equal 1 where plots show the same proportions. Values were transformed to meet the assumption of constant variance, with herpetofaunal and vegetation species cover similarity values transformed with x-square and litter cover group values with the logarithm.

Endpoint difference (ED), a newly developed similarity index (Provencher et al. 1999), is bounded by 0 and 1, and accounts for within-plot variability. ED for treatment i is calculated as:

$$ED_i = \sum n_j \sum \exp[-|X_{ik} - X_{jk}| / \sigma_{eij}] / n_{sp} / N,$$

where the exponential function is of the absolute value of the t statistic, σ_{eij} is the joint standard error of X_{ik} and X_{jk}, n_{sp} is the number of variables, and X is the average value of a single variable. This index is equal to 1 where $X_{ik} - X_{jk} = 0$ for all k, and approaches 1 as the standard error increases. The outer summation in ED_i serves to calculate the weighted average. Transformations to achieve homogeneity of variance were x-square for herpetofaunal similarity values in both years and vegetation species cover in 1998, and square root for litter cover groups in 1997 and 1998 and vegetation species cover in 1997.

When a significant result was calculated for similarity, all 16 similarity values were correlated with each variable's partial similarity contribution to determine which variables explained the similarity pattern. A positive correlation would indicate that the variable supported the similarity pattern; a negative correlation would denote that the variable weakened the pattern. Only significant correlations were retained (≤ -0.482 or ≥ 0.482) (Steel & Torrie 1980). Contributions for proportional similarity were calculated by:

$$1 - 0.5|p_{ik} - p_{jk}|,$$

and averaged per variable over the four reference sites. This was also done for the endpoint difference contribution:

$$\exp[-|Z_{ik} - Z_{jk}| / \sigma_{eij}].$$

Three independent contrasts were selected a priori to answer specific management questions (the maximum allowable number of contrasts is equal to the degrees of freedom for the treatments, or 3, in this case [Sokal & Rohlf 1981]). The first of these paired comparisons compared control to burn plots (C vs. B). This test was selected to examine if the default management technique (prescribed burning) used at EAFB increased the similarity of these plots with the reference, when compared to taking no action (control). Next we compared burn to herbicide plots (B vs. H) to detect differences in similarity between an inexpensive restoration technique (burning) ($12.50/ha in 1995) and a more expensive, labor-intensive treatment (herbicide) ($100/ha). Finally, we wanted to see if there were differences in similarity between the equally expensive ($100/ha) and labor-intensive techniques of herbicide and felling/girdling (H vs. F). These contrasts were tested using the CONTRAST option in SAS.

We examined similarity values for both herpetofaunal and vegetation cover variables using ANOVA. Similarity of tree density variables and overstory cover were not examined due to the deliberate manipulation of this structural component during the restoration effort. Instead, the effect of these manipulations on the tree density was examined only with ANOVA. In several cases, some vegetation cover and herpetofaunal variables had zero values for the majority of the replicates. Where this was the case, only the treatments with sufficient nonzero values were analyzed with ANOVA and contrasts (these are noted on the respective tables).

We predicted that hardwood reduction would increase the similarity of the treatment plots to the reference sites. The herpetofaunal communities in felling/girdling plots should most resemble the reference sites, as a result of oak removal opening the habitat and the structure provided by burnt, fallen, and/or resprouting oaks. With the exception of control plots, herpetofaunal assemblages in burn plots should be least similar to the reference sites, with only 18–41% oak topkill (Provencher et al. 2001*a*). Herbicide plots should result in intermediate similarity values, with high oak mortality, but with dead oaks still standing. However, because fuel reduction burns were conducted recently in felling/girdling and herbicide plots (1997), the reduction of the litter layer may be detrimental for some species. Causes of changes in herpetofaunal abundance, not examined here, could be due to reproduction by individuals within the plots and/or immigration. As previously mentioned, the "Field of Dreams" hypothesis is not explicit about the mechanisms for change. Further, this study was not capable of measuring the role of time lags in herpetofaunal response, although it is conceivable that populations had more time to change in the burn plots than other treatment plots (due to the more recent fuel reduction burns in herbicide and felling/girdling plots).

Results

Tree Density and Overstory Cover

Densities of oaks for all size classes varied among treatments (Table 1). In 1997, densities of small oaks (<5 cm dbh) were not significantly different, due to high variability; however, values were lowest in the burn and herbicide plots and higher in felling/girdling plots that were experiencing resprouting. In 1998, even higher levels of resprouting resulted in significant differences between the felling/girdling plots and the herbicide plots (H vs. F, $p = 0.019$), where again, values were lowest in burn and herbicide plots. Several felling/girdling plots had no medium oaks (5–15 cm dbh) in 1997; however, there was no significant difference among the other treatments. Felling/girdling plots had significantly lower densities of these oaks in 1998, compared to the highest densities in control plots. Again, several felling/girdling plots had no large oaks in 1997 (>15 cm dbh) and herbicide plots also had significantly lower densities than control plots. In 1998, few herbicide plots had large oaks, but there was no significant difference among the other treatments. Generally, oak densities of trees greater than 5 cm dbh were lower or comparable in reference sites than in other treated plots. Densities of smaller oaks (<5 cm dbh) in reference sites, however, were within the range of values found in other plots.

Restoration treatments also had an effect on small (<5 cm dbh) other tree species in 1997, where the majority of the burn plots had zero values, compared to higher values in control plots (Table 1). Reference sites contained more of these trees (<5 cm dbh) than any restoration plot. In 1998, few to no other tree species were found, especially in reference sites. Only one control plot had nonzero values for large pine density, making this treatment biologically different from other treatment plots during 1997. During 1998, there was an experiment-wide difference in the density of large pine trees (>15 cm dbh), indicating higher densities in felling/girdling plots than in control plots. Overall, control plots had low densities of medium and large pines during both years of the study. No significant difference was detected for overstory cover among treatments in 1997, and the difference in 1998 was marginal ($p = 0.060$) (Table 1). Reference sites showed higher overstory cover values than all restoration plots except controls. Differences in tree density results between years, especially for larger trees, may reflect both changes in size class over time, as well as sampling error (i.e., trees on the boundary of the belt transect counted in only one year). Densities of these large trees were low; consequently our sampling area was too small, and the addition or omission of even a single tree could make a large difference between years.

Vegetation Cover Groups

Non-needle (leaf) litter differed among treatments during both years of the study, with significantly more leaf litter in burn plots and by extension in control plots, than in herbicide plots (B vs. H − 1997, $p = 0.002$ and B vs. H − 1998, $p = 0.039$) (Table 2). Reference sites had intermediate values for non-needle litter during 1997 and low values in 1998, compared to the other treatment plots. Herbicide plots had significantly more burned litter than burn plots and control plots (zero values), in 1997 and 1998, the opposite of non-needle litter (B vs. H − 1997, $p = 0.001$ and B vs. H − 1998, $p = 0.008$). Reference sites had very little burned litter in 1997; however, these values greatly increased in 1998 due to wildfires occurring in 2 of the 4 plots.

Herpetofaunal Species and Capture Rates

Species captured during this investigation were representative of the habitat, trap type, season, and proximity to water, although we have no published local sandhill species lists as a reference (Campbell & Christman 1982; Mushinsky 1985; Enge & Marion 1986; Stout et al. 1988; Greenberg et al. 1994*b*; Franz et al. 1995). Twenty-six species were captured over the course of the study (Appendix 1). No significant difference was detected

Table 1. Means, standard errors, and p-values for tree densities (trees/4 × 7.5 m) and overstory cover (%) for restoration (n = 16) and for reference sites (n = 4). P-values are for the overall treatment effect. Italicized letters correspond to the results from testing the three selected independent contrasts (Control vs. Burn, Burn vs. Herbicide, and Herbicide vs. Felling). Different letters indicate a significant difference between treatments. Although selected contrasts may not be significant, a significant overall p-value indicates that at least the most extreme means are different. Reference sites are not part of the experimental design and cannot be tested using ANOVA.

		Treatment					
Tree	*dbh*	*Control*	*Burn*	*Herbicide*	*Felling*	*Reference*	p
1997							
Oak	<5 cm	5.10 ± 1.45	1.35 ± 0.16	1.66 ± 0.39	2.24 ± 1.66	1.44 ± 0.54	0.107
Oak	5–15 cm	1.19 ± 0.37	0.85 ± 0.14	0.85 ± 0.16	0.00 ± 0.00[1]	0.03 ± 0.03	0.956
Oak	>15 cm	0.47 ± 0.09*a*	0.28 ± 0.08*a*	0.10 ± 0.06*a*	0.06 ± 0.06[1]	0.03 ± 0.03	0.039
Pine	<5 cm	0.66 ± 0.31	0.19 ± 0.06	0.72 ± 0.30	0.75 ± 0.33	0.50 ± 0.42	0.471
Pine	5–15 cm	0.07 ± 0.04	0.13 ± 0.07	0.16 ± 0.08	0.16 ± 0.09	0.06 ± 0.06	0.701
Pine	>15 cm	0.13 ± 0.13[2]	0.25 ± 0.11	0.38 ± 0.22	0.41 ± 0.06	0.44 ± 0.12	0.704
Other	<5 cm	1.00 ± 0.41	0.22 ± 0.22[3]	0.13 ± 0.09	0.16 ± 0.09	1.04 ± 0.46	0.072
Other	5–15 cm	0.03 ± 0.03	0.00 ± 0.00	0.06 ± 0.06	0.00 ± 0.00	0.00 ± 0.00	—
Overstory cover		56.23 ± 6.14	48.23 ± 10.61	39.56 ± 4.71	39.72 ± 3.52	56.05 ± 2.34	0.248
1998							
Oak	<5 cm	3.38 ± 0.31*a*	1.13 ± 0.16*a*	0.97 ± 0.24*a*	4.41 ± 1.56b	2.28 ± 0.64	0.046
Oak	5–15 cm	1.59 ± 0.46*a*	0.81 ± 0.28*a*	0.63 ± 0.18*a*	0.16 ± 0.09*a*	0.03 ± 0.03	0.019
Oak	>15 cm	0.22 ± 0.13	0.09 ± 0.06	0.03 ± 0.03[4]	0.13 ± 0.09	0.00 ± 0.00	0.611
Pine	<5 cm	0.66 ± 0.32	0.25 ± 0.09	0.41 ± 0.20	0.69 ± 0.33	0.63 ± 0.50	0.580
Pine	5–15 cm	0.03 ± 0.03[2]	0.25 ± 0.10	0.34 ± 0.13	0.13 ± 0.07	0.19 ± 0.08	0.207
Pine	>15 cm	0.06 ± 0.04*a*	0.19 ± 0.06*a*	0.41 ± 0.17*a*	0.47 ± 0.06*a*	0.41 ± 0.08	0.049
Other	<5 cm	0.44 ± 0.24	0.06 ± 0.04	0.09 ± 0.06	0.19 ± 0.04	0.00 ± 0.00	0.155
Other	5–15 cm	0.03 ± 0.03	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	—
Overstory cover		63.11 ± 4.71	45.00 ± 9.71	40.12 ± 4.46	34.54 ± 5.42	58.17 ± 2.93	0.060

[1]Because the majority of felling/girdling plots had zero values for this variable, ANOVA and contrasts were performed only with control, burn, and herbicide plot values.
[2]Because the majority of control plots had zero values for this variable, ANOVA was performed only with burn, herbicide, and felling/girdling plot values.
[3]Because the majority of burn plots had zero values for this variable, ANOVA was performed only with control, herbicide, and felling/girdling plot values.
[4]Because the majority of herbicide plots had zero values for this variable, ANOVA was performed only with control, burn, and felling/girdling plot values.

for species richness in either year of the study (1997, p = 0.741; 1998, p = 0.319). On average, 9.05 species (± 0.82 SE) were found per plot during 1997 and 8 species (± 0.89 SE) during 1998 (Litt 1999) and species lists for 1997 and 1998 were very similar (Appendix 1). No non-native species were captured.

Capture rates (number of captures/number of trap days) for *Cnemidophorus sexlineatus* (six-lined racerunner) and *Sceloporus undulatus* (eastern fence lizard) varied significantly among treatments during 1997, with higher capture rates for both species in burn plots than in control and herbicide plots (*C. sexlineatus*: C vs. B, p = 0.021; B vs. H, p = 0.023; *S. undulatus*: C vs. B, p = 0.001; B vs. H, p = 0.003) (Table 3). Capture rates of these two species were also high in reference sites in 1997. In 1997, *Bufo quercicus* (oak toad) was rarely, if at all, captured in burn plots (Table 3). During 1998, capture rates for *Tantilla coronata* (southeastern crowned snake), varied significantly among treatments, with higher capture rates found in control plots than in burn plots (p = 0.0015) (and possibly the other treatment plots as well) (Table 3). Reference sites had very low capture rates of this species in 1998, similar to the rate in burn plots. A similar result was found for *Bufo terrestris* (southern toad), where higher capture rates were detected in control plots than burn plots (C vs. B, p = 0.011) (and felling/girdling plots due to zero values). (The same qualitative pattern was seen in 1998 for *Gastrophryne carolinensis*, narrowmouth toad; however, high variability precluded significance) (Table 3). *Anolis carolinensis* (green anole) responded similarly in 1998, where nonzero capture rates were obtained in only one herbicide and felling/girdling plot, making higher capture rates in the control and burn plots biologically significant.

Habitat Heterogeneity

No significant difference was found for habitat heterogeneity values during 1997. All values were fairly high (0.62–0.73) (Table 4), especially in burn and felling/gir-

Table 2. Means, standard errors, and *p*-values for cover groups (%) for restoration ($n = 16$) and for reference sites ($n = 4$). *P*-values are for the overall treatment effect. Italicized letters correspond to the results from testing the three selected independent contrasts (Control vs. Burn, Burn vs. Herbicide, and Herbicide vs. Felling). Different letters indicate a significant difference between treatments. Reference sites are not part of the experimental design and cannot be tested using ANOVA.

	Treatment					
Cover type	*Control*	*Burn*	*Herbicide*	*Felling*	*Reference*	p
1997						
Bare ground	0.02 ± 0.00	0.08 ± 0.01	0.06 ± 0.01	0.07 ± 0.01	0.04 ± 0.01	0.093
Needle litter	0.18 ± 0.03	0.19 ± 0.03	0.35 ± 0.02	0.26 ± 0.04	0.31 ± 0.00	0.293
Non-needle litter	0.51 ± 0.05*a*	0.37 ± 0.03*a*	0.04 ± 0.00*b*	0.10 ± 0.03*b*	0.21 ± 0.01	0.001
Small woody litter	0.03 ± 0.00	0.03 ± 0.00	0.02 ± 0.00	0.02 ± 0.00	0.01 ± 0.00	0.082
Large woody litter	0.01 ± 0.00	0.02 ± 0.00	0.01 ± 0.00	0.02 ± 0.01	0.01 ± 0.00	0.416
Burnt litter	0.00 ± 0.00[1]	0.03 ± 0.01*a*	0.39 ± 0.02*b*	0.30 ± 0.04*b*	0.04 ± 0.01	0.003
1998						
Bare ground	0.03 ± 0.01	0.06 ± 0.01	0.05 ± 0.01	0.07 ± 0.01	0.06 ± 0.01	0.433
Needle litter	0.19 ± 0.03	0.22 ± 0.04	0.29 ± 0.02	0.24 ± 0.03	0.26 ± 0.03	0.635
Non-needle litter	0.48 ± 0.04*a*	0.34 ± 0.02*a*	0.16 ± 0.02*b*	0.20 ± 0.01*b*	0.17 ± 0.02	0.006
Small woody litter	0.03 ± 0.00	0.03 ± 0.00	0.03 ± 0.00	0.02 ± 0.00	0.01 ± 0.00	0.628
Large woody litter	0.02 ± 0.00	0.02 ± 0.00	0.01 ± 0.00	0.02 ± 0.00	0.00 ± 0.00	0.699
Burnt litter	0.00 ± 0.00[1]	0.03 ± 0.00*a*	0.08 ± 0.01*b*	0.08 ± 0.01*b*	0.19 ± 0.06	0.015

[1]Because the majority of control plots had zero values for this variable, ANOVA and contrasts were performed only with burn, herbicide, and felling/girdling plot values.

dling plots, but were also highly variable. A significant treatment effect was found during 1998; however, no contrasts were significant. The contrast between control plots and burn plots showed marginal significance (C vs. B, $p = 0.061$), which would indicate that herbicide and felling/girdling plots should also likely be greater than control plots (untested contrasts), due to higher values and smaller standard errors. Reference sites showed intermediate values in 1998, compared to the other treatment plots.

Table 3. Means, standard errors, and *p*-values for capture rates (number of captures/trap day) of each commonly trapped herpetofaunal species for restoration plots ($n = 16$) and reference sites ($n = 4$). *P*-values are for the overall treatment effect. Italicized letters correspond to the results from testing the three selected independent contrasts (Control vs. Burn, Burn vs. Herbicide, and Herbicide vs. Felling). Different letters indicate a significant difference between treatments. Reference sites are not part of the experimental design and cannot be tested using ANOVA.

	Treatment					
Species	*Control*	*Burn*	*Herbicide*	*Felling*	*Reference*	p
1997						
Bufo quercicus	0.001 ± 0.001	0.000 ± 0.000	0.002 ± 0.001	0.002 ± 0.001	0.001 ± 0.001	0.668[1]
Bufo terrestris	0.005 ± 0.002	0.002 ± 0.001	0.002 ± 0.001	0.003 ± 0.002	0.008 ± 0.005	0.534
Cnemidophorus sexlineatus	0.015 ± 0.008*a*	0.037 ± 0.005*b*	0.016 ± 0.005*c*	0.031 ± 0.009*c*	0.051 ± 0.007	0.046
Gastrophryne carolinensis	0.022 ± 0.010	0.005 ± 0.001	0.012 ± 0.002	0.011 ± 0.001	0.005 ± 0.002	0.193
Sceloporus undulatus	0.002 ± 0.001*a*	0.007 ± 0.001*b*	0.003 ± 0.001*c*	0.003 ± 0.001*c*	0.006 ± 0.001	0.003
Scincella lateralis	0.005 ± 0.002	0.004 ± 0.001	0.002 ± 0.001	0.003 ± 0.001	0.002 ± 0.001	0.316
Tantilla coronata	0.019 ± 0.006	0.011 ± 0.002	0.020 ± 0.006	0.024 ± 0.008	0.009 ± 0.004	0.403
1998						
Anolis carolinensis	0.003 ± 0.002	0.003 ± 0.001	0.000 ± 0.000	0.000 ± 0.000	0.001 ± 0.001	0.914[2]
Bufo terrestris	0.008 ± 0.003*a*	0.000 ± 0.000*b*	0.002 ± 0.001*b*	0.005 ± 0.005	0.003 ± 0.002	0.025[3]
Cnemidophorus sexlineatus	0.013 ± 0.006	0.017 ± 0.002	0.011 ± 0.003	0.019 ± 0.006	0.034 ± 0.006	0.650
Eumeces laticeps	0.002 ± 0.001	0.001 ± 0.001	0.001 ± 0.001	0.001 ± 0.001	0.000 ± 0.000	0.705
Gastrophryne carolinensis	0.005 ± 0.003	0.003 ± 0.001	0.001 ± 0.001	0.001 ± 0.000	0.001 ± 0.001	0.106
Sceloporus undulatus	0.002 ± 0.001	0.006 ± 0.002	0.001 ± 0.001	0.004 ± 0.002	0.002 ± 0.001	0.101
Scincella lateralis	0.002 ± 0.001	0.002 ± 0.000	0.001 ± 0.000	0.002 ± 0.001	0.002 ± 0.001	0.508
Tantilla coronata	0.014 ± 0.002*a*	0.004 ± 0.001*b*	0.005 ± 0.001*b*	0.007 ± 0.001*b*	0.004 ± 0.002	0.007

[1]Because the majority of burn plot values were zero, only control, herbicide, and felling/girdling plot values were analyzed using ANOVA.
[2]Because the majority of herbicide and felling/girdling plot values were zero, only control and burn plot values were analyzed.
[3]Because the majority of felling/girdling plot values were zero, only control, burn, and herbicide plot values were analyzed.

Table 4. Means, standard errors, and p-values for habitat heterogeneity (evenness) values for restoration (n = 16) and reference sites (n = 4). P-values are for the overall treatment effect. Italicized letters correspond to the results from testing the three selected independent contrasts (Control vs. Burn, Burn vs. Herbicide, and Herbicide vs. Felling). Different letters indicate a significant difference between treatments. Although selected contrasts may not be significant, a significant overall p-value indicates that at least the most extreme means are different. Reference sites are not part of the experimental design and cannot be tested using ANOVA.

	Treatment					
Year	*Control*	*Burn*	*Herbicide*	*Felling*	*Reference*	p
1997	0.62 ± 0.10	0.73 ± 0.02	0.65 ± 0.01	0.71 ± 0.06	0.72 ± 0.01	0.567
1998	0.58 ± 0.06*a*	0.70 ± 0.04*a*	0.77 ± 0.01*a*	0.77 ± 0.03*a*	0.66 ± 0.02	0.020

Similarity Indices

Proportional Similarity. In 1997 no differences were detected in the proportional similarity to the reference condition for litter cover (Table 5). The slightly lower values in herbicide and felling/girdling plots reflect the recent hot fires (due to heavy fuels created by the treatments). Differences were significant in 1998, with higher similarity values detected in burn plots (and herbicide and by extension felling/girdling plots) than in control plots (C vs. B, p = 0.043). Non-needle litter, small and large woody litter, and burnt litter contributed most to these differences. No difference was detected in the similarity of vegetation species cover in either year (Table 5).

Strong differences in proportional similarity for the herpetofauna were found during 1997 (Table 5). Herpetofaunal assemblages in burn and felling/girdling plots were significantly more similar to the reference condition than those found in control (C vs. B, p = 0.002) and herbicide plots (H vs. F, p = 0.001). The strongest contributors to this difference were *C. sexlineatus*, *G. carolinensis*, *Eumeces laticeps* (broadhead skink), *S. undulatus*, and *Eumeces egregius* (mole skink). A similar pattern resulted in 1998, that is, treated plots had slightly higher similarity values than the control, but high levels of variation, especially in control plots, precluded significance (see Discussion).

Endpoint Difference. No significant difference was found for endpoint difference for litter cover groups in either year (Table 5). As with proportional similarity, end-

Table 5. Means, standard errors, and p-values for similarity indices of litter cover, vegetation species cover, and herpetofaunal capture rates using both metrics to compare restoration (n = 16) with reference sites (n = 4). P-values are for the overall treatment effect. Italicized letters correspond to the results from testing the three selected independent contrasts (Control vs. Burn, Burn vs. Herbicide, and Herbicide vs. Felling). Different letters indicate a significant difference between treatments.

	Treatment				
Variable	*Control*	*Burn*	*Herbicide*	*Felling*	p
		Proportional similarity			
Litter					
1997	0.68 ± 0.09	0.72 ± 0.07	0.59 ± 0.04	0.59 ± 0.06	0.430
1998	0.53 ± 0.05*a*	0.61 ± 0.04*b*	0.68 ± 0.01*b*	0.67 ± 0.02*b*	0.045
Vegetation					
1997	0.47 ± 0.02	0.43 ± 0.02	0.43 ± 0.02	0.53 ± 0.04	0.107
1998	0.49 ± 0.02	0.45 ± 0.02	0.55 ± 0.04	0.56 ± 0.05	0.199
Herpetofauna					
1997	0.49 ± 0.09*a*	0.76 ± 0.03*b*	0.49 ± 0.04*c*	0.64 ± 0.06*d*	0.004
1998	0.46 ± 0.12	0.64 ± 0.04	0.67 ± 0.06	0.65 ± 0.07	0.308
		Endpoint difference			
Litter					
1997	0.28 ± 0.05	0.31 ± 0.08	0.29 ± 0.03	0.20 ± 0.04	0.434
1998	0.12 ± 0.02	0.21 ± 0.06	0.17 ± 0.03	0.17 ± 0.03	0.371
Vegetation					
1997	0.30 ± 0.02	0.30 ± 0.02	0.24 ± 0.01	0.30 ± 0.02	0.061
1998	0.49 ± 0.02	0.45 ± 0.02	0.55 ± 0.04	0.56 ± 0.05	0.239
Herpetofauna					
1997	0.32 ± 0.02*a*	0.40 ± 0.01*b*	0.31 ± 0.01*c*	0.33 ± 0.00*c*	0.009
1998	0.34 ± 0.03	0.40 ± 0.02	0.39 ± 0.02	0.38 ± 0.01	0.307

point difference values for vegetation species cover were not significant (Table 5).

During 1997, differences in herpetofaunal similarity values were significant, with burn plots showing higher similarity to the reference sites than the other plots (C vs. B, $p = 0.005$; B vs. H, $p = 0.002$; H vs. F, $p = 0.447$) (Table 5). *G. carolinensis*, *A. carolinensis*, *T. coronata*, *Storeria occipitomaculata* (redbelly snake), *S. undulatus*, and *C. sexlineatus* contributed most to the differences found. As with proportional similarity, treated plots were again slightly more similar to reference sites than control plots in 1998; however, no significant differences were detected due to variability among replicates.

Discussion

Changes in hardwood densities and litter cover groups demonstrate that the restoration treatments are indeed modifying the habitat structure (see Provencher et al. 2001*a*; Provencher et al. 2001*b*). These changes in habitat structure may have altered the herpetofaunal community. According to the "Field of Dreams" hypothesis, we expected that the degree of structural change (i.e., hardwood reduction) would result in a condition for the vegetation and therefore the herpetofauna that most closely matched the reference sites (i.e., felling/girdling would be the most similar, followed by herbicide, burn, and control plots). Instead a slightly different ranking resulted. Using proportional similarity, we determined that during 1997 the herpetofaunal community in burn and felling/girdling treatment plots was more similar to the reference sites than it was to herbicide and control plots. Using endpoint difference, we determined that burn plots were significantly more similar to the reference sites in 1997. In 1998 plots that had experienced some type of fire (spring burn or fuel reduction) were more similar to the reference condition than to the control plots for both indices, although high variability prevented achieving significance. These results suggest that fire increased similarity of the herpetofaunal community to the reference condition. Fire may provide habitat heterogeneity as a result of the characteristic patchy nature of this disturbance. The pattern of habitat heterogeneity values closely matched that of proportional similarity in both years; however, high variability precluded achieving significance in 1997. Variability in habitat heterogeneity values was always highest and means were always the lowest in control plots, further emphasizing the role of fire in creating habitat patchiness. Patchiness from spring fire may result from variation in fuel loads or indirectly from the partial topkill of hardwoods. Fire is also known to indirectly stimulate the herbivorous arthropods through new vegetation growth (Nagel 1973; Harris & Whitcombe 1974). The degree of hardwood kill may not be the most important factor (e.g., burn vs. felling/girdling or herbicide application) in these open woodlands. Mushinsky and Gibson (1991) suggested that management practices that create a habitat mosaic will likely benefit the highest number of species in Florida sandhills. In their study of silvicultural practices on herpetofauna in scrub habitats, Greenberg et al. (1994*b*) found that providing similar conditions to those seen after disturbance was the most important factor in maintaining typical reptile communities, regardless of the method.

The high variability seen in the control plots in 1998 was likely due to a severe drought during most of the sampling season, which magnified the differences among the four control plots. Average total rainfall for the sampling period was 283.5 mm and 187.8 mm (from late May through July) in 1997 and 1998, respectively (Litt 1999). Two of the control plots were drier and in 1997 contained fewer amphibians and litter-loving species than the other control plots, whereas the other two control plots may have served as refuges for these species. These differences increased in 1998 due to the lack of rainfall, when even fewer individuals of these species (e.g., *G. carolinensis* and *B. terrestris*) were captured in two of the control plots, though they were captured more often in the other two plots. This caused widely varying and increasing similarity values, especially for proportional similarity. Treated plots were not significantly different from controls in 1998, possibly due to high variability in a drought year.

Higher capture rates of both *C. sexlineatus* and *S. undulatus* in burn plots than in herbicide and control plots in 1997 may indicate that the degree of habitat heterogeneity may play a role in herpetofaunal community composition. *C. sexlineatus* is a rapidly moving, active-foraging species, and has been found to prefer open microhabitat areas with mineral soil (Allen & Neill 1953; Hardy 1962; Conant & Collins 1998). Woody litter and trunks are preferred microhabitat for *S. undulatus*, a sit-and-wait predator (Allen & Neill 1953). If both of these microhabitat characteristics are present in burn plots, where capture rates of these two species were highest, the degree of habitat heterogeneity present may be at an optimal level to satisfy both species. Herbicide treatment followed by fuel reduction burns and no treatment in control plots may represent the extremes, ranging from an acute elimination of understory cover to a large accumulation of litter. Neither extreme appears to be favorable for these species, where capture rates in the felling/girdling plots were not significantly different from herbicide plots. Similar results were observed in 1998, albeit with a high level of variability. In contrast, *T. coronata*, which feeds on insect larvae (Semlitsch et al. 1981), seemed to favor the presence of cover and the litter layer in control plots, where it reached its highest capture rates in 1998. This result also was found by Franz et al. (1995) with *Tantilla relicta* (peninsular crowned snake) in peninsular Florida, showing a preference for thin to moderate ground cover. Franz et al. (1995) also found less use of areas with a thick litter layer, suggesting that there may be some threshold level of litter cover required by this species. Greenberg et al. (1994*b*) found no difference in numbers of *T. relicta* in scrub habitats with fire or under different silvicultural site-preparation techniques. However, there was some evidence of fewer *T. relicta* in stands with high levels of woody debris. The importance of litter cover could have been particularly exaggerated in 1998 resulting in significant differences among treatments, again due to the severe drought experienced in this area. These conditions may have affected other species as well (e.g., *B. terrestris* and *A. carolinensis*).

The species that contributed most to differences in proportional similarity and endpoint difference found in 1997 fall into two groups. The first, *C. sexlineatus* and *S. undulatus*, represent species that are likely indicators of fire-maintained areas. The second group consists of species that are more indicative of fire suppression, requiring some kind of litter for cover (e.g., amphibians or semi-fossorial species) or foraging (e.g., species that utilize chemoreception for prey detection; Mushinsky 1992). Reaching a consensus using multiple similarity indices is important because each index provides slightly different information (e.g., endpoint difference incorporates within-plot variability) and can produce slightly different outcomes (e.g., contrasts for herpetofaunal similarity in 1997). Both indices identified *C. sexlineatus*, *G. carolinensis*, and *S. undulatus* as potential indicators. These species could be evaluated for use in a monitoring program, at EAFB or elsewhere, if power analysis were conducted to determine samples sizes required for the proposed monitoring sampling design (e.g., sample sizes needed to detect a predetermined percentage of change in the capture rates of a rarely captured species may be too large to be incorporated feasibly into a monitoring program).

Conclusions

We have shown that fire increases the similarity of the herpetofaunal community in burned restoration plots to reference sites. If management objectives require quick midstory reduction or if smoke management problems prevent prescribed burning, then using more expensive methods may be necessary. Herbicide rapidly kills hardwoods with no resprouting, while felling/girdling quickly removes midstory hardwoods, but with a high level of resprouting. Because only a portion of the herpetofaunal community was sampled during this investigation, the impact of herbicides on possibly sensitive herpetofaunal species (e.g., *Rana okaloosae*, Florida bog frog), especially in adjacent habitat types, is not well known. Therefore, this treatment should be applied with caution, and monitoring efforts should also be undertaken to document possible effects. Fuel-reduction burns following felling/girdling and herbicide treatments are necessary to reduce fire hazards from heavy fuel loads resulting from these treatments and to control oak resprouting in felling/girdling plots. Fire has the added benefit of stimulating new plant growth. If spring fire is not feasible, these other methods provide adequate alternatives, although examination of the long-term effects of these treatments, as well as recurring fire (a 3–5 year burn interval), are warranted.

We believe that these data address an important restoration question: how do sandhill reptiles and amphibians respond to restoration techniques that are being widely used throughout the southeastern United States? The spatial scale of this experiment, large replicated treatment plots, and the use of high-quality reference sites as the restoration goal provide ideal conditions for measuring restoration success. Restoration of longleaf pine sandhill remnants is a conservation priority in the Southeast. In fact, northwest Florida has been identified as the sixth most important diversity hotspot in North America (Stein et al. 2000). Therefore, it is imperative that we understand the response of these communities to management actions that are actively being employed.

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Appendix. List of all species captured during sampling in each year. During 1997, there were 16,576 trap days and 1,251 herpetofaunal captures. In 1998, 23,696 trap days resulted in 976 captures.

Species	*Common Name*	*1997*	*1998*
Anurans			
Bufo quercicus	oak toad	√	√
Bufo terrestris	southern toad	√	√
Gastrophryne carolinensis	eastern narrowmouth toad	√	√
Hyla cinerea	green tree frog		√
Rana clamitans	bronze frog	√	√
Rana utricularia	southern leopard frog		√
Scaphiopus holbrookii	eastern spadefoot toad	√	
Salamanders			
Notophthalmus viridescens	central/spotted newt	√	
Plethodon grobmani	southeastern slimy salamander	√	√
Pseudotriton ruber	red salamander	√	
Lizards			
Anolis carolinensis	green anole	√	√
Cnemidophorus sexlineatus	six-lined racerunner	√	√
Eumeces egregius	mole skink	√	√
Eumeces laticeps	broadhead skink	√	√
Sceloporus undulatus	eastern fence lizard	√	√
Scincella lateralis	ground skink	√	√
Snakes			
Cemophora coccinea	scarlet snake	√	√
Coluber constrictor	black racer	√	√
Diadophis punctatus	ringneck snake	√	√
Lampropeltis triangulum	scarlet kingsnake	√	√
Micrurus fulvius	coral snake		√
Nerodia fasciata	banded water snake	√	
Sistrurus miliarius	pigmy rattlesnake	√	√
Storeria occipitomaculata	red-bellied snake	√	
Tantilla coronata	southeastern crowned snake	√	√
Virginia valeriae	smooth earth snake	√	√