SHORTLEAF PINE–BLUESTEM RESTORATION FOR RED-COCKADED WOODPECKERS IN THE OUACHITA MOUNTAINS: IMPLICATIONS FOR OTHER TAXA

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Abstract: The more xeric south- and west-facing slopes of the Ouachita Mountains of west-central Arkansas once supported fire-maintained shortleaf pine (*Pinus echinata*) forests with a well-developed herbaceous understory. Fire suppression following the original harvest of these forests resulted in forests with increasingly abundant woody vegetation in the understory, midstory, and canopy, and a very suppressed herbaceous understory. Due largely to these habitat changes, red-cockaded woodpecker (*Picoides borealis*) populations declined to extremely low levels by the 1990s. Ouachita National Forest managers increased the emphasis on prescribed burning and thinning in the late 1970s and initiated landscape-level restoration of shortleaf pine-bluestem (*Schizachyrium* spp. and *Andropogon* spp.) communities in 1996, in part to support recovery of the endangered red-cockaded woodpecker. Restoration involves thinning the pine and hardwood overstory, mechanical removal of midstory pines and hardwoods, and prescribed burning on approximately a 3-year return interval. This management (which will eventually encompass over 48,000 ha) has been very successful in returning large areas to a condition similar to historical photographs of original shortleaf pine forests. Restoration results in a more open, pine-dominated overstory and an increase in herbaceous vegetation compared to untreated control areas. The abundance of nectar resources was significantly higher in treated areas compared to controls, peaking in the first growing season following burning and declining thereafter. Overall abundance and species richness of butterflies, amphibians, reptiles, mammals, and birds were generally greater in restored areas compared to controls. Many species of conservation concern, including Diana fritillary (*Speyeria diana*), red-cockaded woodpecker, northern bobwhite (*Colinus virginianus*), prairie warbler (*Dendroica discolor*), brown-headed nuthatch (*Sitta pusilla*), and Bachman's sparrow (*Aimophila aestivalis*), also responded positively to restoration. Of the taxonomic groups studied, few maintained higher abundances in the control areas. Overall, regional abundance and natural patterns of diversity of the examined taxa will be enhanced with restoration of shortleaf pine–bluestem communities on appropriate sites in the Ouachita Mountains.

Key words: amphibians, Arkansas, butterflies, fire, Interior Highlands, moths, nectar sources, *Picoides borealis*, *Pinus echinata*, reptiles.

A substantial component of public lands (>800,000 ha on U.S. Forest Service lands alone) in the southeastern U.S. are now being managed for the endangered red-cockaded woodpecker (Bowman et al. 1999). This basically single-species management approach is appropriate given the species' status as both an endangered and a keystone species (Conner et al. 2001*a*). However, this management is also good ecosystem management and restores these areas to a closer approximation of pre-European settlement conditions (Conner et al. 2001*a*, Masters et al. 2002). Consequently, it is important to understand the impacts of this management on non-target species, as these public lands will be increasingly important as regional reserves for sustaining biodiversity (Brennan et al. 1995). Knowledge of the impacts of this management on other species of conservation concern is of special interest.

The Ouachita Mountains physiographic region of west-central Arkansas and southeastern Oklahoma encompasses an area of 3,237,600 ha consisting predominantly of east-west oriented ridges and valleys with elevations ranging from 150 to 820 m (Bukenhofer and Hedrick 1997). Historically, many of the forests of this region (especially those on the more xeric south- and west-facing aspects) burned on a regular basis by fires ignited by lightning and Native Americans (Foti and Glenn 1991, Masters et al. 1995). These frequently burned forests had open, pine-dominated overstories, sparse midstories, and a highly diverse understory of grasses and forbs (Featherstonhaugh 1844, du Pratz 1975, Nuttall 1980, Foti and Glenn 1991, Masters et al. 1995) (Figure 1), which provided suitable habitat for elk

Figure 1. Photograph of shortleaf pine-bluestem habitat in the Ouachita Mountains, Arkansas made in the early 1900s prior to timber harvest and major alteration of the fire regime. Photo courtesy of the Ouachita National Forest.

(*Cervus elaphus*), bison (*Bison bison*), white-tailed deer (*Odocoileus virginianus*), and the red-cockaded woodpecker (Jansma and Jansma 1991, Smith and Neal 1991). Without frequent burning, these fire-maintained forests may succeed to an oak (*Quercus* spp.)-hickory (*Carya* spp.) community (Neal and Montague 1991).

Although the Ouachita Mountain landscape is still predominantly forested, forest structure and composition have changed dramatically due primarily to logging of the original forests and suppression of fire (Bukenhofer and Hedrick 1997). Generally, stands are now younger and denser, with abundant woody midstories, dense pine-hardwood overstories, and reduced herbaceous vegetation (Fenwood et al. 1984, Masters 1991, Sparks 1996) (Figure 2a). Elk and bison are extirpated from this region and populations of many species associated with forests and frequent fire, such as Bachman's sparrow and brown-headed nuthatch, have declined (Jackson 1988).

The once common red-cockaded woodpecker is currently endangered throughout the southeastern United States, including the Ouachita Mountains (Neal and Montague 1991), due primarily to incompatible timber harvesting practices (e.g., clearcutting coupled with short rotations) and alteration of the fire regime (Conner et al. 2001*a*). The red-cockaded woodpecker population on the Ouachita National Forest (ONF) declined to a low of 11 groups by 1996 (Rudolph et al. 2004*a*). Despite this low population size, the ONF was selected as 1 of 11 national forests designated for red-cockaded woodpecker recovery (U.S. Forest Service 1996*a*). This designation was based on the large land base available for habitat restoration and the decision to recover red-cockaded woodpecker populations in each of the major ecological regions within their historic range. A population goal of 250 potential breeding groups of red-cockaded woodpeckers has been set for the ONF (Bukenhofer et al. 1994, U.S. Forest Service 1996*b*).

In 1979, largely in response to declining red-cockaded woodpecker populations, the ONF increased its emphasis on prescribed burning and thinning. In July 1996, they amended their Forest Plan to gradually restore 48,706 ha (7.3% of the Forest) to a shortleaf pine-bluestem grass ecosystem (U.S. Forest Service 1996*b*: Appendix G-3). The primary goals of this initiative are to recover red-cockaded woodpecker populations and to restore the historic shortleaf pine-bluestem grass ecosystem within Scott and Polk Counties, Arkansas, on the western portion of the ONF (U.S. Forest Service 1996*a*, Bukenhofer and Hedrick 1997).

Restoration is achieved by wildlife stand improvements (WSI, thinning the overstory and removing most of the midstory) followed by burning the stands approximately every 3 years. This burning interval favors grasses and forbs and reduces midstory and understory woody plants (Masters 1991, Masters et al. 1996). The rotation age for shortleaf pine forest types also has been increased from 70 to a minimum of 120 years. Rotation age was lengthened to increase the availability of older pines that are required by red-cockaded woodpeckers (Rudolph and Conner 1991) and to increase the incidence of red heart fungus (*Phellinus pini*) (Bukenhofer and Hedrick 1997) that facilitates cavity excavation by red-cockaded woodpeckers. Stand regeneration will be accomplished mainly through the use of irregular shelterwood and seed tree cuts; a portion

Fig 2a

Figure 2. Photographs of (a) un-restored stand (WSI-B2) and (b) restored stand in the Ouachita Mountains, Arkansas.

Fig 2b

of the overstory will be retained indefinitely under both of these regeneration methods.

Restoration efforts have been remarkably successful. Visual examination of the restored stands (Figure 2b) reveals a major change in stand structure compared to stands that have not yet been restored (Figure 2a). More importantly, restored stands resemble descriptions and photographs of pine forests in the region prior to the initial logging and alteration of the pre-European fire regime (Figure 1).

Studies have been conducted in the Ouachita Mountains to evaluate the impacts of pine-bluestem restoration on vegetation (Sparks 1996; Sparks et al. 1998, 1999), white-tailed deer forage production (Masters et al. 1996), small mammals (Masters et al. 1998), northern bobwhite (Cram et al. 2002), and breeding bird communities (Wilson 1994, Wilson et al. 1995, Masters et al. 2002). We initiated studies to evaluate the responses to pine-bluestem restoration efforts in the Ouachita Mountains of amphibians and reptiles in 1999, butterflies and nectar resources in 2000, and moths in 2001. Here we summarize preliminary data from this ongoing research along with published findings for birds and small mammals. Although our primary focus is on community level responses (relative abundance, species richness, and diversity), we also present findings for several species of conservation concern.

METHODS

Study Areas and Treatments

Wilson et al. (1995) conducted breeding bird surveys in 6 untreated (control) stands, 6 unburned WSI stands (WSI-NB), 9 WSI stands within the first year following spring burning (WSI-B1), 6 WSI stands within the second year following spring burning (WSI-B2), and 6 WSI stands within the third year following spring burning (WSI-B3). Their burned stands had been through the 3-year burning cycles only 1-4 times; at least half of their WSI-B1 stands were WSI-NB during the previous sampling season (Wilson et al. 1995). Small mammals were live trapped in a subset of these same stands (3 stands per treatment) in 1992 and 1993 (Masters et al. 1998). All their stands were located on the Poteau, Mena, and Cold Springs Districts of the ONF and ranged in size from 14 to 45 ha.

Our research on the responses of amphibians, reptiles, butterflies, moths, and nectar resources to shortleaf pine-bluestem restoration was conducted on the Poteau District (34º45'N, 94º15'W) of the ONF in west-central Arkansas. The soils of this area, derived primarily from shale and sandstone parent material, are generally shallow, rocky, and drought prone. The site index for shortleaf pine on these soils is about 18 m at 50 years (Guldin et al. 1994, western physiographic block). The climate is characterized by warm and humid summers, mild winters, and an average annual rainfall of about 132 cm (Skiles 1981). The more xeric south-facing slopes tend to be dominated by shortleaf pine, whereas the more mesic north-facing slopes tend to be dominated by oaks, hickories, and other hardwoods (Foti and Glenn 1991).

We sampled 9 restored stands (3 each of the 1-, 2-, and 3-year-old burns; henceforth WSI-B1, WSI-B2, and WSI-B3) and 3 unrestored (control) pine-hardwood stands. In successive years, WSI-B1 stands become WSI-B2 stands, WSI-B2 stands become WSI-B3 stands, and WSI-B3 stands are subjected to prescribed burning in the winter or spring and become WSI-B1 stands. With 1 exception, all of our restored stands had undergone a minimum of 4 burning rotations prior to their inclusion in this study, and all but 2 of the 9 restored sites had been burned 5-7 times since the late 1970s (Table 1). Although district burning records did not always indicate burning dates, at least 4 of the 9 restored stands had been burned late in the growing season (September and October, Table 1). With the exception of these burns, all burns for which the date was known were conducted between January and April.

The sizes of the treated stands in which our surveys were conducted varied from 10.5 to 42.1 ha and averaged 22.7 ha (Table 1). However, all of the treated stands were part of larger burning units (ranging from 64.8 to 1,335.5 ha since 1999) that were burned the same day as our survey stands (W.G. Montague, Ouachita National Forest, unpublished data). Additional areas adjacent to these larger burning units were burned on other days within the same burn years for 5 of 9 treated areas. Our 12 stands were generally rectangular in shape with slopes of <20%. Control stands were located in the general vicinity of the restoration areas and averaged 32.0 ha in size (Table 1). Control stands in all studies reported here had a history of timber management and were not representative of old-growth or pre-settlement conditions; however, they were representative of pre-restoration conditions currently prevalent on much of the ONF.

Without fire, herbicides, or mechanical treatments, thinning alone (the WSI-NB treatment shown in Tables 2 and 3) will not maintain restored conditions or suitable red-cockaded woodpecker habitat (Conner et al. 2001*a*). Thus, we did not include this treatment in our experimental design.

Late rotation stands on south-facing slopes on the western part of the ONF, comparable to our control stands, typically average about 23.4 m²/ha of conifer basal area (99.3% shortleaf pine and 0.7% eastern

Table 1. Compartment and stand number, stand size, year of wildlife stand improvement (WSI), and burning history for 3 control and 9 pine-bluestem restoration areas on the Poteau District of the Ouachita National Forest.

Treatment[a]	Compartment	Stand	Size (ha)	WSI Year	No. of Burns[b]	Dates Burned[c]
Control	1277	13	27.9	NA	NA	NA
	1276	1	36.4	NA	NA	NA
	1294	2	31.6	NA	NA	NA
WSI-B1	1259	11	12.1	1990	6	3/00, 3/97, 3/96, 9/94, 4/92, 3/89, 3/78
	1257	16	17.8	1990	5	3/00, 3/97, 9/95, 3/92, 3/89, 3/78
	1281	18	19.4	1984	4	3/00, 4/97, 3/93, 3/90, 1987
WSI-B2	1265	6,12[d]	42.1	1980	7	4/02, 4/99, 10/95, 3/92, 3/89, 1986, 4/83, 1980
	1253	5,8[d]	41.3	1980/83[e]	6	4/02, 4/99, 2/96, 3/90, 2/87, 9/84, 2/81
	1272	7	19.0	1986	5	4/02, 3/99, 4/96, 3/93, 3/89, 1984
WSI-B3	1274	12,19[d]	24.3	1989	3	3/01, 3/98, 9/94, 3/92
	1274	20	17.8	1989	7	3/01, 3/98, 10/95, 3/92, 3/89, 2/84, 1983, 1/81
	1274	21	10.5	1989	6	3/01, 3/98, 3/95, 3/92, 3/89, 2/84, 1/81

[a]Burning designations (B1, B2, and B3) are for 2000. WSI-B1 = wildlife stand improvement, first growing season following burn. B2 = second growing season following burn. B3 = third growing season following burn.
[b]Number of prescribed burns between initiation of pine-bluestem restoration efforts in the late 1970s and the beginning of this study.
[c]Dates without an indicated month were most likely dormant season burns; none of the control stands had been burned or thinned since at least 1978 (W.G. Montague, Ouachita National Forest, personal communication).
[d]Sampling area consisted of 2 smaller stands that were combined into 1 treatment area.
[e]Stands 5 and 8 were thinned in 1980 and 1983, respectively.

redcedar, *Juniperus virginiana*) and 7.4 m^2/ha of hardwood basal area, for a combined total of 30.8 m^2/ha (Guldin et al. 1994). Residual basal area following WSI treatments is typically about 13.7-16.1 m^2/ha for pines and 1.4-1.6 m^2/ha for hardwoods; thus, roughly 36% of the pine and 80% of the hardwood basal area is typically removed during WSI treatments (W.G. Montague, Ouachita National Forest, personal communication). However, additional hardwood basal area may be retained whenever stand or landscape levels fall below U.S. Forest Service guidelines (L.D. Hedrick, Ouachita National Forest, personal communication).

Data Collection

Birds in each stand were surveyed once by 3 observers on different dates each year within 6 40-m-radius point count plots spaced 130-150 m apart within each stand. Each plot was sampled for 8 min between 0600 and 1100 from 13 to 21 May 1992 and 1993 (Wilson et al. 1995). Although these authors resampled breeding birds on a subset of these areas in 1999 and 2000 (Masters et al. 2002), their results did not include relative abundance, richness, and diversity values across all 4 years and they did not provide orthogonal comparisons for controls versus burning treatments. Therefore, we only present bird data for the first 2 years of this study (Wilson et al. 1995).

Small mammals were live trapped for 7 consecutive days between 27 December and 4 January in 1992 and 1993 at 80 trap stations in each stand. Trap stations were located at 15-m intervals along 3-8 (depending on stand shape and size) randomly located transects. One Sherman live trap (7.6 x 8.9 x 22.9 cm) was set at each station (Masters et al. 1998). A number of habitat measures, including percent canopy cover, conifer basal area, hardwood basal area, herbaceous plant production, and woody plant production, were made during the bird and small mammal studies (Wilson et al. 1995, Masters et al. 1998).

Amphibians and reptiles were sampled using drift fence and funnel trap arrays with supplemental pitfall traps. Funnel traps consisted of a 1.2- x 1.2- x 0.46-m high box with a funnel entrance on each side. The bottoms and tops were constructed of plywood, and the sides and funnels were constructed of 3.2-mm mesh hardware cloth. The top contained an access door. A water source was provided in each trap. Four 15-m long drift fences extended perpendicularly from the sides of each trap at the midpoint of each funnel entrance. Drift fences were constructed of 3.2-mm mesh hardware cloth 61 cm in width buried approximately 10-15 cm in the ground. A pitfall trap, consisting of an 18.9-l plastic bucket, was buried flush with the soil surface at the distal end of each drift fence. A 50- x 50-cm plywood cover was placed over each pitfall bucket, raised approximately 10-15 cm above the lip of the bucket, to provide shade and deflect precipitation. A small amount of leaf litter was placed in each bucket to provide cover and a microhabitat for captured animals.

Three trap arrays were installed in each of the 12 stands. Individual arrays were placed in a triangular pattern with ≥150 m separation. Arrays were placed ≥50 m from edge habitat, i.e., stand boundaries, roads, etc., and ≥75 m from permanent and intermittent ponds and streams. Trap arrays were checked weekly for 24 weeks (early April to late September) during all 3 years (1999-2001). All captured vertebrates were identified to species, recorded, and released approximately 50 m from the trap.

We measured habitat in September and early October at 4 points located 7 m beyond the distal end of each drift fence. Canopy closure was measured with a spherical densiometer and conifer and hardwood basal areas were determined using a 1-factor metric prism. Ocular estimates of percent cover of herbaceous monocots, herbaceous dicots, and woody understory vegetation <2 m high were made within 3 1- x 1-m subplots adjacent to the above measurement points; ocular estimates were made to the nearest percent ≤10% and to the nearest 5% above 10%. Values for the 3 subplots were averaged prior to analysis. Canopy cover and overstory tree basal areas were measured in 1999, and understory cover (herbaceous monocots and dicots and woody plant cover) was measured in 2000. Because overstory structure was only minimally affected by fire treatments, it was not remeasured in 2000.

Lepidoptera were sampled in the same control and treatment stands used in the amphibian and reptile study. Adult butterflies (Papilionoidea and Hesperioidea) were censused using a time-constrained walking census along a 500-m transect (Pollard 1977, Gall 1985). These transects were located along the sides of the triangle formed by the drift fence arrays. Individual transects were censused by slowly walking the length of the transect in approximately 20 min. Time required to count butterfly aggregations or to occasionally net and identify individuals was not included in the 20-min period. Most individuals (87.2%) were identified to species; unidentified individuals were recorded to the lowest taxonomic category possible.

Butterfly census counts were conducted 4 times per year (first week of April, June, August, and September) beginning in 2000; although these counts are scheduled to extend through 2002, we only present data for 2000. Census counts were replicated 3 times (a different observer on 3 separate days) during each monthly survey period. Individual censuses were conducted between 0900 and 1330 hrs CST on days when temperatures were between 18 and 36º C, and wind velocity beneath the canopy was not too strong to suppress butterfly flight (Beaufort Scale ≤4). Censusing was restricted to periods when sunlight was sufficient to cast discernible shadows. During periods of partial cloud cover, censusing was temporarily halted. The response of butterflies to light, wind, and cloud cover varies seasonally and daily in complex ways. Consequently, observer judgment further constrained censusing to those periods when butterfly activity appeared to be substantial.

Moths were sampled using UV light traps consisting of a 12-V fluorescent tube, powered by a 12-V battery, mounted over a 13.2-l plastic bucket. Two plastic bottles containing ethyl acetate and sponge wicks were placed in each bucket to kill captured arthropods. During each night of sampling, light traps were operated via electronic timer for a total of 8 hrs, consisting of 4 periods of 2 hrs duration equally spaced between 30 min after sunset and 30 min before sunrise.

A single light trap was placed in each stand on top of 1 of the funnel traps used in the amphibian and reptile surveys. Light traps were operated for 3 consecutive nights, during the first week of each sampling month (April, June, August, and September) when heavy rain was not forecast. Additional nights were occasionally required when equipment malfunctioned, when excessive precipitation depressed number of captures, or when water damage prevented moth identification.

The contents of the light trap buckets were collected each morning and sorted immediately, or frozen for later processing. Contents were initially sorted into macrolepidoptera, microlepidoptera, and other arthropods. Only information on macrolepidoptera and a few species of larger microlepidoptera is reported at this time. Macrolepidoptera and larger microlepidoptera were identified to species or morphospecies, and 1 or more vouchers were pinned and labeled for each taxon.

Nectar resources were quantified during each lepidopteran sampling month within 3 1- x 100-m belt transects located parallel to each butterfly census transect. All nectar resources were counted and recorded by species. For most species, individual flowers or composite heads (capitula) were enumerated. Inflorescences, or portions thereof, were counted for a few species with small and/or dense aggregations of

Table 2. Stand characteristics for several Ouachita National Forest studies.

Study[a]	Parameter	Control	WSI-NB	WSI-B1	WSI-B2	WSI-B3	P
Wilson et al. 1995	Canopy cover (%)	84.1A[b]	65.1B	67.9B	70.4B	69.3B	0.001
	Conifer BA[c] (m^2/ha)	19.5A	13.5C	14.8BC	17.3AB	16.8BC	0.005
	Hardwood BA (m^2/ha)	6.9A	4.4AB	3.9B	2.6B	2.9B	0.045
	Total BA (m^2/ha)	26.4A	17.9B	18.7B	20.0B	19.7B	0.002
Masters et al. 1996	Herbage production (kg/ha)[d]	77	399	361	437	383	NA[e]
	Woody plant production (kg/ha)[d]	82C	589A	356B	486AB	544A	0.001
Current study	Canopy cover (%)	94.1A	NA	71.1BC	79.7B	62.6C	<0.001
	Conifer BA (m^2/ha)	18.1	NA	16.4	19.4	15.9	0.220
	Hardwood BA (m^2/ha)	5.0	NA	3.2	3.2	2.2	0.077
	Herbaceous monocot cover (%)	7.6A	NA	22.6AB	29.0B	23.9AB	0.021
	Herbaceous dicot cover (%)	1.1A	NA	20.2B	13.2C	5.1A	<0.001
	Woody understory cover (%)	20.8A	NA	18.2A	28.7AB	36.3B	0.017

[a]The bird and small mammal data reported by Wilson et al (1995) and Masters et al. (1998) were obtained from the same study areas. Thus, the Wilson et al. (1995) data characterize habitat for both of their studies.

[b]Row means followed by the same letter were not different, $P > 0.05$ (LSD). The current study used a 1-way ANOVA with subsampling, REGWQ (SAS Inst. Inc. 1988:598), and the same alpha level. The overstory and understory data shown for the current study were collected in 1999 and 2000, respectively.

[c]Basal area.

[d]Current-year growth measured in late July and early August.

[e]Masters et al. (1996) did not test for differences in total herbage production among treatments.

flowers (e.g., *Ceanothus americanus, Allium* spp., *Solidago* spp., Apiaceae). Enumeration decisions were generally based on the structure that most closely approximated a separate landing site for a typical butterfly.

Data Analysis

Except for bird relative abundance and diversity, we present data on birds and small mammals from Wilson et al. (1995) and Masters et al. (1998) without further analyses. We computed total relative bird abundance using data from Wilson et al. (1995, Table 2). Bird diversity values were provided by R.E. Masters (Tall Timbers Research Station, unpublished data).

Relative abundances for amphibians, reptiles, nectar resources, butterflies, and moths include data for unknown species; however, data for unknowns were excluded from species richness and diversity indices. Diversity within each of these groups was computed using Shannon's diversity index (Shannon and Weaver 1998).

Amphibian and reptile data were analyzed separately for each of the 3 years of our study. Abundance was calculated as the number of captures/100 trap days (n = 504 trap days/stand/year). Data were summed for each stand and year and divided by 5.04 to obtain captures/100 trap days. Data were rank transformed prior to 1-way analysis of variance (Kruskal-Wallis ANOVA) and REGWQ comparisons at $P < 0.05$ (SAS Inst. Inc. 1988:598). Species richness is the total number of species encountered in each stand each year, averaged across the 3 replications of each treatment.

Nectar resource abundance was calculated as the number of flowers, inflorescences, and composite heads summed across transects (300 m^2) within each stand and season. Species richness is the total number of species encountered in each stand, by month, averaged across the 3 replicates of each treatment.

To derive monthly relative abundances for butterflies and moths, total counts for each stand were averaged across observers (3) or trap nights (3), respectively, and these means were then averaged across the 3 replications to obtain treatment means. Because abundance varied greatly within treatments, data were rank transformed prior to analyzing in a 1-way ANOVA with REGWQ at $P < 0.1$ (SAS Inst. Inc. 1988:598). Butterfly species richness, the total number of species encountered in each stand each month, was averaged across the 3 replicates of each treatment. Because we have yet to identify all the moths to species or morphospecies, we do not present moth richness and diversity data at this time.

RESULTS

Stand and Understory Conditions

In 1993 and 1994, Wilson et al. (1995) found that herbage production was 4.7 to 5.7 times higher in treated stands than in controls (Table 2). Woody plant

Table 3. Relative abundance, species richness, and diversity of birds and small mammals from untreated controls, thinned but unburned (WSI-NB), and thinned pine-bluestem habitat within the first (WSI-B1), second (WSI-B2), and third (WSI-B3) year following prescribed burning in the Ouachita Mountains of west-central Arkansas. Diversity values were computed using Shannon and Weaver (1998).

Group	Parameter	Year	Controls	WSI-NB	WSI-B1	WSI-B2	WSI-B3	*P*
Birds[a]	Relative abundance	1992-93	237.4	314.7	356.0	371.0	336.7	0.037
	Richness	1992-93	13.2	15.2	15.2	16.5	15.3	0.399
	Diversity	1992-93	2.13	2.26	2.21	2.24	2.14	0.854
Mammals[b]	Relative abundance	1992	2.2^{C}	7.4^{A}	7.1^{AB}	4.3^{BC}	3.9^{C}	0.007
		1993	0.9	3.9	4.2	2.3	2.6	0.069
	Richness	1992	2.7	4.7	3.3	3.7	3.0	0.447
		1993	1.3^{C}	1.7^{BC}	4.3^{A}	2.7^{AB}	3.0^{AB}	0.009
	Diversity	1992	1.1	1.5	0.9	1.4	1.1	0.837
		1993	0.3^{B}	0.3^{B}	1.6^{A}	0.9^{AB}	1.1^{A}	0.017

[a]Data from Wilson et al. (1995). Relative abundance (individuals/40 ha) values were computed from the means provided in their Table 2. R. E. Masters (Tall Timbers Research Station) provided the bird diversity values. The probability value for relative abundance was provided in their text, but the authors did not provide multiple range test results.
[b]Data from Masters et al. (1998). Relative abundance is small mammal captures/100 trap nights.

production was lowest on the controls and highest on the WSI-NB; within the 3 restored treatments, woody plant production increased each year following burning.

Canopy cover and conifer and hardwood basal areas were presumably much lower in our restored than control stands immediately after WSI treatments were imposed, as they were in the earlier study (Wilson et al. 1995). However, considerable time had passed since our restored stands had been thinned, during which substantial diameter growth had occurred. Thus, while canopy cover remained lower ($P < 0.0001$) in restored stands than controls, there were no longer significant differences in pine or hardwood basal areas between the restored and control stands by 1999.

By the time we measured understory conditions in September and early October, woody plant cover below 2 m in the WSI-B1 and WSI-B2 stands was comparable ($P > 0.05$) to the controls. That these differences were not significant is at least partially due to substantial within treatment variability, as the WSI-B2 treatment averaged 38.0% higher woody cover than the controls (Table 2). However, by the third year post burning, woody cover was greater ($P < 0.05$) in WSI-B3 stands than in the controls.

Percent cover of herbaceous monocots and dicots was 3.0 to 3.8 and 4.6 to 18.4 times greater, respectively, in treated stands than in controls. Within restored treatments, monocot cover was highest the second year following burning. Herbaceous dicot cover was significantly higher on the WSI-B1 and WSI-B2 compared with controls, but had declined to levels comparable to the controls by late in the third year following burning. Herbaceous dicot cover declined significantly each year following burning, presumably due to expansion of woody plants.

On comparable sites within the Ouachita Mountains, Masters et al. (1996) found that wildlife stand improvement cuts without burning (WSI-NB) increased legume production in late July and early August by 118% (11 vs. 24 kg/ha) and increased forb production by 621% (14 vs. 101 kg/ha) compared to untreated controls. They found no difference ($P > 0.05$) in legume production among the WSI-B1 (62 kg/ha), WSI-B2 (72 kg/ha), and WSI-B3 (67 kg/ha) treatments, nor between the WSI-NB and WSI-B1. However, the WSI-B2 and WSI-B3 treatments produced significantly more legumes than the WSI-NB. Thus, burning increased legume production above that obtained by thinning alone. Forb production within all 4 WSI treatments was significantly higher than in the controls; however, there were no differences ($P > 0.05$) among the WSI-NB (101 kg/ha), WSI-B1 (149 kg/ha), WSI-B2 (124 kg/ha), and the WSI-B3 (110 kg/ha) treatments (Masters et al. 1996).

Birds

Abundance of birds in the 3 restored treatments was 41.8 to 56.3% higher than in the controls (Table 3) (Wilson et al. 1995). Within the 3 restored treatments, abundances peaked during the second year post burning and then declined in the third year. Avian species richness values in the 3 restored treatments were slightly higher than in the controls, but none of these differences were significant ($P = 0.399$) (Table 3). Species diversity values were also comparable ($P = 0.854$) across all treatments. That richness and diversity were similar across all treatments is due primarily to species replacement, rather than an overall change in number of species in response to habitat alterations associated with restoration. Based on nesting guilds, the 3 restored treatments contained significantly more shrub nesting birds than control stands, and controls had fewer canopy nesting species than WSI-B1 and WSI-B2 stands (Wilson et al. 1995).

The restoration treatment benefited a number of bird species associated with pine-grasslands. Of the 10 of these species encountered by Wilson et al. (1995), 4 (Bachman's sparrow; brown-headed nuthatch; red-cockaded woodpecker; and red-headed woodpecker, *Melanerpes erythrocephalus*) were never encountered within untreated controls (Wilson et al. 1995; Table 3), but were present within all restored treatments. Five additional species associated with pine-grasslands (chipping sparrow, *Spizella passerina*; indigo bunting, *Passerina cyanea*; northern bobwhite; prairie warbler; and eastern wood-pewee, *Contopus virens*) that were infrequently encountered in controls had significantly higher frequencies of occurrence in 1 or more of the restored treatments (Wilson et al. 1995). Cram et al. (2002) found that relative abundance of northern bobwhites averaged 9.4 to 19.2 times higher in WSI-B2 and WSI-B3 stands, respectively, than controls; WSI-B1 stands had 5.5 times higher relative abundance values for northern bobwhite than controls, but this difference was not statistically significant. All 6 of the pine-grassland associates (Bachman's sparrow, northern bobwhite, prairie warbler, brown-headed nuthatch, red-cockaded woodpecker, and red-headed woodpecker) that are species of conservation concern within the Ozark/Ouachita Physiographic Area (Fitzgerald and

Pashley 2000) seem to have benefited from the restoration efforts (Wilson et al. 1995, Masters et al. 2002).

Of the remaining 38 bird species encountered, only 3 (ovenbird, *Seiurus aurocapillus*; black and white warbler, *Mniotilta varia*; and whip-poor-will, *Caprimulgus vociferus*) had higher ($P < 0.05$) densities within controls than within restored stands (Wilson et al. 1995). Ovenbirds and whip-poor-wills are both species of conservation concern within this region (Fitzgerald and Pashley 2000).

Small Mammals

Relative abundance of small mammals (as measured by capture rates) was consistently higher in the WSI-NB and the burning treatments than in the controls during both years of the Masters et al. (1998) study. However, these differences were only significant the first year of their study. Masters et al. (1998) attributed these increases in relative abundance more to the effect of thinning than to fire. In 1992 there was no difference in small mammal capture rates between the WSI-NB and WSI-B1, but both of these had significantly higher capture rates than the controls (Table 3). In 1992, capture rates in the WSI-B2 and WSI-B3 were 2.6 and 1.8 times higher, respectively, than in the controls, but these differences were not significant ($P > 0.05$). During 1993, small mammal captures rates in the restored treatments were 2.6 to 4.7 times higher than in the controls, but none of the second-year differences were significant.

There were no significant differences in small mammal species richness or diversity during 1992. However, during 1993, the observed increases in species richness and diversity were attributed to both thinning and fire effects (Masters et al. 1998). Differences in species richness within the 3 restored treatments were not significant in 1993 (Table 3). However, the 3 burning treatments had 2.1 to 3.3 times higher species richness than the controls and these differences were statistically significant in 1993. No species were adversely affected by the WSI or burning (Masters et al. 1998). Thus, pine-bluestem restoration had mostly beneficial (or neutral) effects on the small mammal community.

Amphibians

There were few significant differences in overall amphibian relative abundance, species richness or diversity during any year, presumably because of high within treatment variability (Table 4). Amphibian species richness and diversity were significantly different only in 1 of the 3 years. In most years, overall amphibian relative abundance, richness, and diversity were comparable to or higher in the restoration treatments than in the controls; where they were lower for a given burn age, they did not remain so for long. During

Table 4. Mean relative abundance (captures/100 trap days), species richness, and diversity of amphibians and reptiles in untreated late rotation pine-hardwood stands and restored pine-bluestem sites during the first, second, and third year following prescribed burning in the Ouachita Mountains, Arkansas. Three drift fence arrays were sampled for 24 weeks on each of 3 sites per treatment from early April to late September for 1999 to 2001.

			Control		WSI-B1		WSI-B2		WSI-B3		
Group	Parameter	Year	$\bar{x}^a$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	P^b
Amphibians	Abundance	1999	17.9	7.3	39.6	19.7	13.3	0.9	10.9	3.6	0.346
		2000	9.3	2.4	17.8	2.0	10.6	3.1	6.5	3.3	0.075
		2001	7.8	3.1	10.3	2.0	7.3	2.3	13.0	7.5	0.731
	Richness	1999	9.7	2.0	9.3	2.8	8.0	0.6	9.0	1.7	0.979
		2000	6.3^A	0.7	9.0^B	0.0	9.0^B	0.6	6.0^A	1.0	0.008
		2001	7.7	1.2	9.3	0.7	7.7	0.9	7.3	1.3	0.546
	Diversity	1999	1.31^A	0.22	0.85^B	0.11	0.91^B	0.04	1.40^A	0.08	0.006
		2000	1.15	0.12	1.23	0.09	1.52	0.05	1.34	0.16	0.106
		2001	1.67	0.16	1.60	0.15	1.66	0.12	1.51	0.07	0.773
Reptiles	Abundance	1999	12.2	1.2	20.5	3.9	23.7	0.8	17.7	3.6	0.095
		2000	9.8^A	0.4	14.2^B	1.7	13.4^{AB}	1.7	16.7^B	0.3	0.021
		2001	9.8	2.6	11.8	1.4	10.3	1.5	11.1	0.3	0.941
	Richness	1999	14.0	1.5	18.0	3.5	20.3	1.8	18.3	2.9	0.348
		2000	14.0	0.6	17.3	0.9	16.0	1.5	16.7	0.3	0.107
		2001	13.3	2.3	16.0	1.2	17.7	1.2	16.3	1.2	0.523
	Diversity	1999	2.23	0.10	2.23	0.14	2.37	0.16	2.45	0.09	0.569
		2000	2.22^A	0.04	2.53^B	0.05	2.30^{AB}	0.05	2.37^{AB}	0.08	0.034
		2001	2.17	0.11	2.33	0.08	2.46	0.05	2.33	0.11	0.201

[a]Means within rows followed by the same letter are not statistically different ($P > 0.05$) using REGWQ (SAS Inst. Inc. 1988:598).
[b]Kruskal-Wallis ANOVA probability.

2000, amphibian species richness was about 50% higher in WSI-B1 and WSI-B2 stands than in WSI-B3 and control stands. However, in 1999, amphibian diversity was significantly lower in WSI-B1 stands compared to controls.

Even within subgroups (frogs, toads, or salamanders), within treatment variation was high enough to mask possible treatment differences. There was no consistent pattern in frog abundance across years and there were no statistical differences among treatments in any year (Figure 3). Frog captures were highest in the WSI-B1 during 1999 and 2000, but highest on the WSI-B3 during 2001. Toad captures followed fairly consistent patterns across treatments in 1999 and 2000, but treatment means were statistically different only in 2000 when more toads were captured in the WSI-B1 than WSI-B3 stands (Figure 3). Toad captures were generally about 3 times higher in WSI-B1 stands than the controls. Too few salamanders were captured to draw any conclusions except that yearly variability in capture rates was greatest in control stands (Figure 3).

Reptiles

As with amphibians, there was considerable within treatment variability in the reptile data. Consequently, the only significant treatment differences were for relative abundance and diversity in 2000 (Table 4). In 2000, reptile capture rates were higher ($P < 0.05$) in first- and third-year burns compared with controls and reptile diversity was higher in first-year burns than in controls. Although differences were not statistically significant, in all but 1 case (1999 diversity for first-year burns) overall reptile relative abundance, richness, and diversity values were numerically higher on the 3 restored treatments than the controls. These overall reptile community response patterns were also generally reflected in the snake, lizard, and turtle capture rates (Figure 3). Capture rates of snakes tended to increase as the burns matured. In 2000, capture rates of snakes were significantly higher in WSI-B3 stands than WSI-B2 and control stands. Capture rates of lizards also tended to be lowest in control stands, but 1999 was the only year when significantly more lizards were caught in WSI-B1 and WSI-B2 stands than the other 2 treatments (Figure 3). Although capture rates of turtles tended to be lowest on controls, turtles were captured too infrequently to permit definitive conclusions.

Nectar Resources

The abundance of nectar resources was significantly different across treatments for all months (Table 5, Figure 4a). Within months, abundance of nectar resources was consistently lowest in controls, with the single exception of the WSI-B1 treatment in April immediately following the prescribed fire treatment. Nectar resource abundance generally declined with each successive year post burn. The WSI-B1 and WSI-B2 treatments produced significantly greater nectar resources than the controls, except for the WSI-B1 treatment in April immediately following the prescribed burn. With declining nectar resource abundance in WSI-B3 treatments, significant differences between WSI-B3 and controls were only detected in June, although numerical abundance was still greater in WSI-B3 treatments.

Species richness data exhibited a similar pattern in all months, although significant differences were only detected in June and October (Table 5). Generally, species richness was lowest in controls and highest in WSI-B1 treatments, with progressive declines in WSI-B2 and WSI-B3 treatments. The single exception was the WSI-B1 treatment in April following the recent prescribed fire treatments. Species diversity data were consistent with the pattern of lowest values in the controls and highest values in the 3 burn treatments within each month, although significant differences were not detected in any month (Table 5).

Butterflies

The relative abundances of adult butterflies (Papilionoidea and Hesperioidea) were significantly different across treatments in all months (Figure 4b, Table 5). Patterns of abundance were generally consistent across months: controls had the lowest census totals in all 4 sample months and WSI-B1 treatments generally had the highest totals. The only exception was April, when the surveys were conducted only 7-29 days post burn, i.e., prior to substantial re-growth of the vegetation. Significant differences among treatments within months (Figure 4b) were broadly consistent, exhibiting a pattern of lowest relative abundance of butterflies in controls and higher relative abundance in treatments, with abundance decreasing in successive years postburn. One species of conservation concern, the Diana fritillary, was significantly more abundant in restored than control stands, apparently responding to the increased abundance of high quality nectar

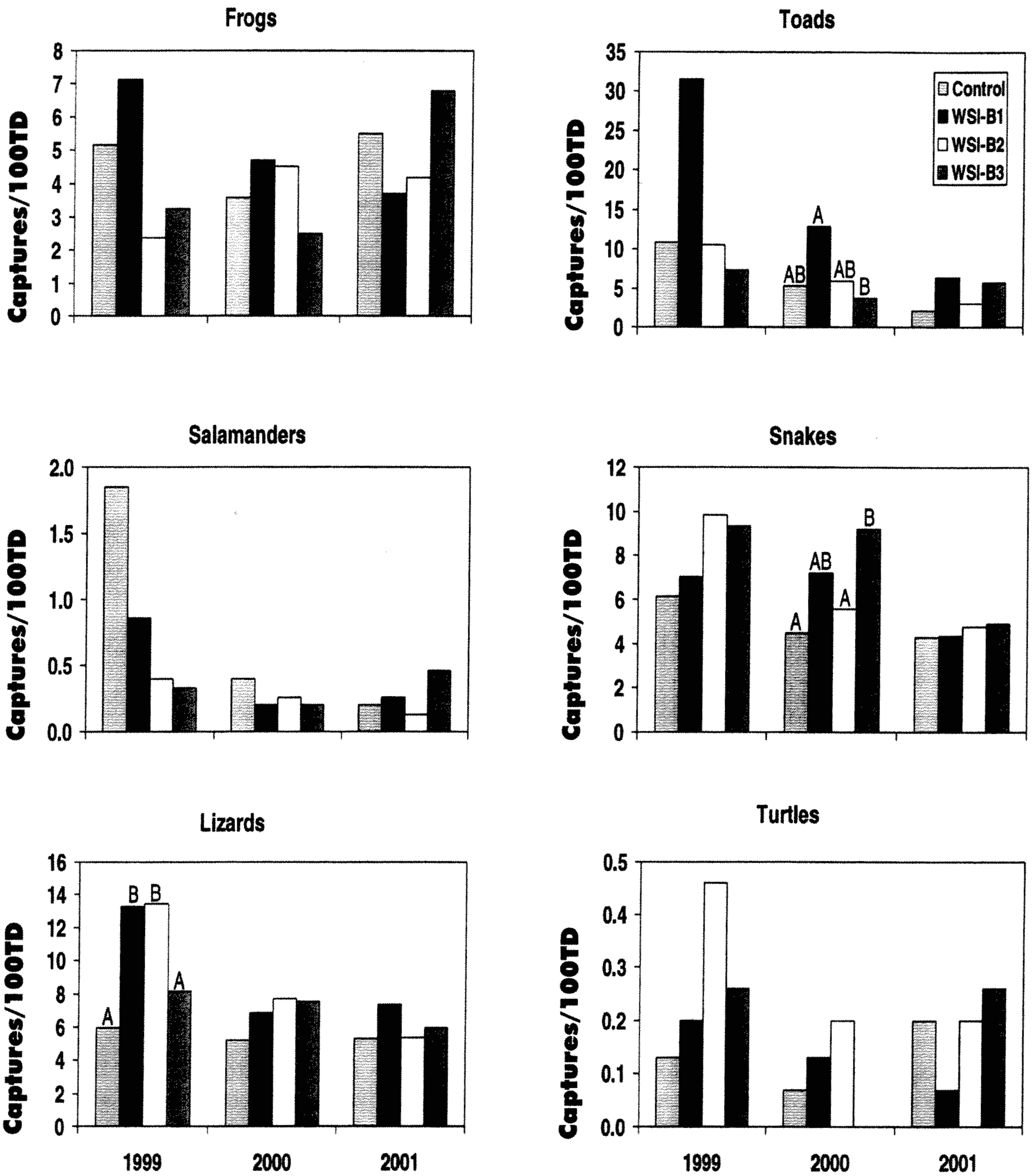

Figure 3. Relative abundance (captures/100 trap days) of amphibian and reptile groups in unrestored late rotation pine-hardwood stands and restored pine-bluestem sites during the first, second, and third year following prescribed burning in the Ouachita Mountains, Arkansas.

Table 5. Nectar plant and lepidoptera responses to pine-bluestem restoration in the Ouachita Mountains of Arkansas. Butterfly and nectar source data are for 2000; moth data are for 2001.

Group / Parameter	Month	Controls $\bar{x}$	Controls SE	WSI-B1 $\bar{x}$	WSI-B1 SE	WSI-B2 $\bar{x}$	WSI-B2 SE	WSI-B3 $\bar{x}$	WSI-B3 SE	*P*
Nectar										
Relative abundance[a]	Apr	150.7AB	12.4	84.7A	45.5	960.0B	532.1	154.7AB	62.3	0.048
	Jun	49.7A	15.4	1,018.3B	364	420.3B	134.9	399.3B	172.3	0.002
	Aug	3.7A	0.7	417.7B	157.3	583.0B	312.6	56.3A	41.4	0.002
	Oct	7.3A	5.1	746.3C	66.4	84.3B	45.9	59.3AB	21.9	<0.001
Richness	Apr	3.3	0.3	1.7	0.9	4.3	0.9	4.7	1.3	0.185
	Jun	4.7A	0.9	14.0B	1.5	11.3B	1.3	11.3B	1.7	0.005
	Aug	2.3	1.3	7.3	2.4	7.7	1.8	3.7	1.7	0.120
	Oct	0.7A	0.3	8.3C	1.2	3.0B	1.2	2.3AB	0.3	0.001
Diversity	Apr	0.36	0.07	0.71	0.21	0.60	0.11	0.56	0.20	0.428
	Jun	1.12	0.17	1.25	0.14	1.51	0.17	1.31	0.26	0.613
	Aug	0.54	0.54	1.25	0.26	1.00	0.28	0.66	0.34	0.576
	Oct	0.00	0.00	0.77	0.11	0.67	0.34	0.75	0.13	0.202
Butterflies										
Relative abundance[b]	Apr	12.1A	2.7	13.7A	5.3	32.1B	5.9	36.6B	3.1	0.028
	Jun	21.2A	7.8	68.4B	26.2	54.8B	11.4	26.9A	1.8	0.024
	Aug	7.4A	2.1	58.9B	14.1	44.7B	8.9	11.6A	2.3	<0.001
	Oct	1.6A	0.3	16.4B	1.7	6.8C	1.1	7.6C	0.9	<0.001
Richness	Apr	7.3A	0.3	12.7B	2.4	13.0B	0.6	16.7B	3.2	0.046
	Jun	5.0A	1.5	21.7B	2.7	19.3B	0.7	13.3B	1.7	<0.001
	Aug	6.7A	1.7	16.3B	2.8	17.0B	4.1	10.3AB	1.9	0.046
	Oct	3.3	0.9	7.7	1.2	4.7	1.2	5.0	1.5	0.207
Diversity	Apr	1.61A	0.02	2.23B	0.16	2.11B	0.08	2.27B	0.08	0.035
	Jun	1.05A	0.25	2.44B	0.01	2.38B	0.07	1.73C	0.23	<0.001
	Aug	1.63	0.18	1.82	0.20	1.92	0.36	2.08	0.09	0.658
	Oct	0.66	0.38	1.36	0.11	1.22	0.26	0.76	0.21	0.371
Moths										
Relative abundance[c]	Apr	142.0A	22.6	34.0B	5.4	84.3AB	15.7	81.4B	14.4	0.023
	Jun	348.2A	18.0	202.3B	25.1	454.0A	110.1	388.6A	59.3	0.041
	Aug	100.8A	8.7	207.4B	25.9	220.0B	22.4	170.2B	16.0	0.006
	Oct	70.0	18.0	139.1	27.1	133.6	21.9	112.7	19.6	0.194

[a]Number of nectar resources per 300 m^2 (see text for explanation).
[b]Mean number per 500 m transect.
[c]Mean number per trap night.

resources following prescribed fire (D. C. Rudolph et al., Southern Research Station, unpublished data).

Species richness values were significantly different in all months except October (Table 5), and the pattern in all months was similar to that of the abundance data. In all months, richness was lowest in the controls and significantly lower for most comparisons. There were also consistent taxonomic differences between controls and restored treatments. Specifically, Satyrinae (satyrs and wood nymphs), species that rarely take nectar and typically fly in shaded habitats, were numerically dominant in control stands whereas other taxonomic groups were dominant in the treatment stands (D. C. Rudolph, Southern Research Station, personal observation). Species diversity followed similar patterns, with lower diversity in the controls; however, these numerically large differences were only significant in 2 of the 4 months due to high variability in the data (Table 5).

Moths

Relative abundances of moths were significantly different for all sampling months except October (Table 5, Figure 4c). The pattern of differences across treatments was similar to that of the butterflies for August and October, showing higher relative abundances in restored stands than in controls. However, the pattern for April was the reverse, with highest relative abundance in the control stands. In June, relative abundance values were inconsistent with later months, presumably because abundances were depressed due to residual effects of recent burning in the WSI-B1 stands. Moth abundances seem to be responding to the restoration treatments, but in complex ways that will require more detailed analysis of additional years of data to decipher.

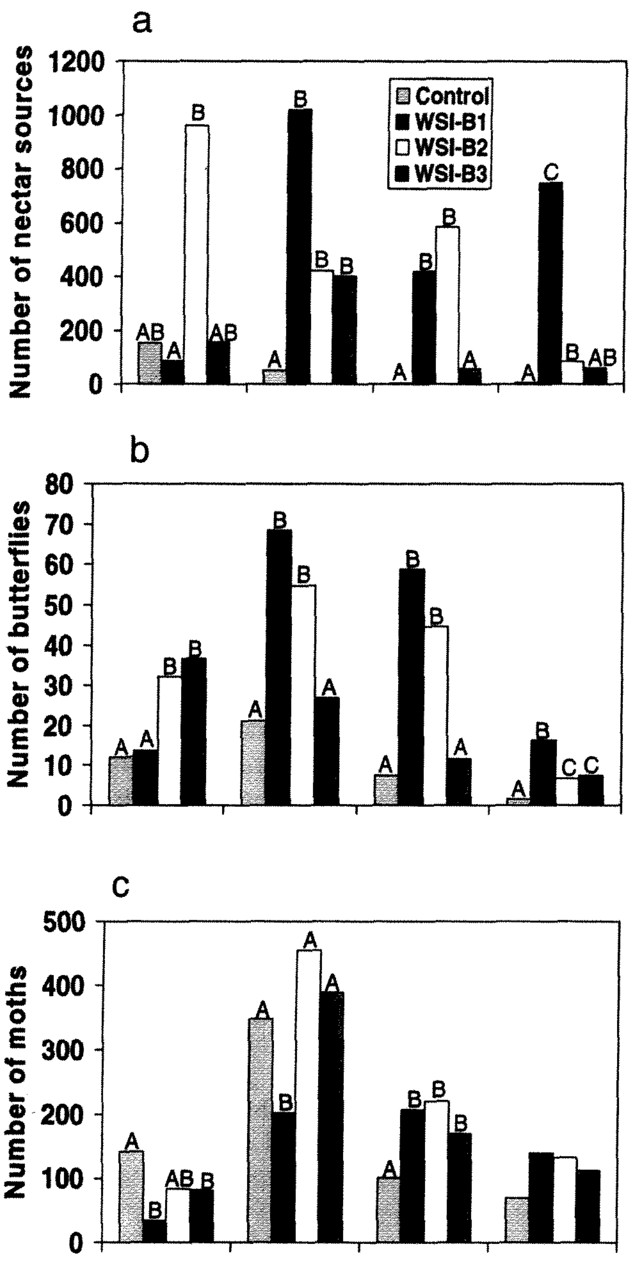

Figure 4. Relative abundance of (a) nectar resources (number/300 m^2), (b) butterflies (mean number/500-m transect), and (c) moths (mean number/trap night) in control and restored shortleaf pine-bluestem stands in April, June, August, and October during the first, second, and third year following prescribed burning in the Ouachita Mountains, Arkansas.

DISCUSSION AND CONCLUSIONS

According to Foti and Glenn (1991), Native Americans in this region modified the burning regime by increasing fire frequency, reducing the intensity of burns, or shifting the timing to later in the autumn when the fire's ability to kill woody vegetation was reduced. Pine-bluestem restoration efforts in the Ouachita Mountains currently involve predominantly March and April burns. For the 46 prescribed burns with known burning dates (Table 1), burns on our 9 restored areas occurred during March (56.6%), April (19.6%), February (8.7%), September (8.7%), October (4.3%), and January (2.2%). Numbers of present-day lightning-set fires are highest in August followed by September and then July (Foti and Glenn 1991); few lightning-set fires occur in March, April, and February, which is when our study areas were most often burned. Thus, present-day prescribed burning practices for pine-bluestem restoration poorly match seasonal patterns of historic lightning or anthropogenic fires (Sparks et al. 1998). The U.S. Forest Service recognizes this disparity but will likely find it difficult to greatly increase their growing-season burning program due to potential resource damage (e.g., overstory scorch), more hazardous burning conditions, and other concerns (Haines et al. 2001). Nevertheless, late dormant-season burns (Mar and Apr) were more effective on these sites than late growing-season burns (Sep and Oct) in reducing woody understory sprouting, which was accompanied by higher herbaceous species abundance and richness (Sparks et al. 1998, Sparks et al. 2002).

Due to efficient fire suppression over many decades, the changes in vegetation structure on the ONF have been dramatic. Experience on the ONF, and in other localities in the southeastern United States (Waldrop et al. 1992), demonstrate that prescribed fire alone is not capable of rapidly achieving restoration objectives. Consequently, on many forests where rapid restoration of habitat for red-cockaded woodpecker recovery is essential, mechanical (WSI in this paper) or chemical means are frequently used in combination with an aggressive prescribed fire regime (Conner et al. 2001*a*).

In our study, herbaceous dicot cover declined significantly each year following burning. By the third year post burning, rapidly expanding woody understory, litter accumulation, and other factors had suppressed herbaceous monocots and dicots. Because many of these herbaceous species serve as nectar sources for lepidoptera, frequent burning may be necessary to achieve sustained, abundant herbaceous nectar resources.

Hardwood management practices on the ONF have been a contentious issue for decades. Consequently, it is not surprising that the pine-bluestem efforts have met with some opposition from those concerned about hardwood composition and hard mast supplies for deer and other wildlife. Because of these concerns, the ONF implemented 2 Forest-wide standards addressing hardwood composition. The first of these is a landscape objective to have ≥20% of each compartment in mast-producing capability, which is defined as hardwood or hardwood-pine forest types with trees ≥50 years old. The second standard is a within-stand objective to maintain 10-30% hardwood basal area, where possible, within each managed pine stand (U.S. Forest Service 1990*a*). While red-cockaded woodpeckers are generally intolerant of hardwoods adjacent to roost trees, we believe this hardwood retention goal will not adversely impact the woodpecker recovery efforts underway there.

Many of the north-facing slopes throughout the ONF are already being managed as either hardwood or hardwood-pine stands. In addition, unharvested strips (greenbelts) are retained along all ephemeral, intermittent, and permanent streams for watershed protection when the adjacent stands are thinned or regenerated. Because of the rugged terrain and relatively high rainfall, greenbelts and streamside zones comprise a substantial portion of the landscape throughout the ONF, including pine-bluestem restoration areas. Where within-stand hard mast supplies are inadequate, greenbelts and portions of streamside zones could be managed for increased mast production along with other wildlife habitat features that may be in short supply (Thill et al. 1994). Thus, opportunities exist for increasing hard mast supplies within pine-bluestem restoration sites as well as the rest of the ONF. Additionally, it is important to remember that shortleaf pine-bluestem restoration is only planned for 7.3% of the ONF.

When pine-bluestem restoration efforts were initiated on the ONF, most treated stands (including presumably our 9 stands) were thinned to a residual pine basal area of about 16 m^2/ha. More recently, the residual pine basal area target has been lowered to 14 m^2/ha. We suspect this 12.5% lower initial basal area would not have changed our results appreciably.

Restoration of shortleaf pine-bluestem communities has been remarkably successful where implemented on the ONF. Thinning of canopy trees and removal of most midstory trees (WSI) followed by prescribed fire on a 3-year rotation has resulted in a vegetation structure that approximates that known from early accounts (Bukenhofer and Hedrick 1997), and closely resembles historic photographs made prior to the initial harvests and major alterations of the fire regime (Figures 1 and 2b). The primary similarities, in contrast to stands that have not been restored (Figure 2a), are reduced canopy closure, greatly reduced woody midstory, and a well-developed herbaceous understory. Measures of vegetation structure made in conjunction with the studies reported above support these patterns.

Changes in relative abundances of vertebrate taxa were the most apparent response to restoration in the current studies. Measures of abundance for reptiles, mammals, and birds were consistently higher for all restored treatments than for controls. In the case of amphibians, abundances in the first year post-burn and 4 out of 6 comparisons in later years were higher than controls. Variability was quite high, however, and not all of these comparisons were statistically significant. Changes in species richness and diversity are less apparent due primarily to species replacements, rather than an overall change in number of species in response to habitat alteration associated with restoration.

These results are consistent with other studies of vertebrate responses to habitat management undertaken to restore red-cockaded woodpecker habitat (Brennan et al. 1995, Burger et al. 1998, Provencher et al. 2002*a*). A large proportion of vertebrate species of conservation concern that are present, or potentially present in pine-dominated habitats in the Ouachita Mountains, also benefit from restoration management. Such species include the red-cockaded woodpecker (Conner et al. 2001*a*), Bachman's sparrow (Plentovich et al. 1998), and additional species discussed above.

Concerns have been expressed about the negative effects of fire on some species of lepidoptera (Dana 1991; Swengel 1996, 1998), generally in association with breakdown of metapopulation structure due to habitat fragmentation. Within less fragmented landscapes, fire has been recognized as an important factor in maintaining suitable habitat for individual species (Williams 1995, Kwilosz and Knutson 1999) and faunas (Swengel 1996, Rudolph and Ely 2000). The rapid reoccupation of sites by adult lepidoptera, following prescribed fire, in our studies suggests that breakdown of metapopulation structure is not occurring.

Our study of Papilionoidea and Hesperioidea demonstrates substantial increases in relative butterfly abundance in fire-maintained habitat compared to control areas. The overall pattern also demonstrates a decline in numbers detected during the second and third growing seasons following early spring prescribed fires. These results are similar to those obtained in other areas where reduction of woody vegetation by fire, or other means, results in increases in the number of adults detected (Swengel 1996, Rudolph and Ely 2000).

The increases in butterfly detections parallel substantial, but highly variable, increases in nectar resources available following restoration of shortleaf pine-bluestem habitat. Flower abundance in the understory peaks in the first or second growing season following spring burning, and declines thereafter. Concurrent declines in butterfly detections and nectar resources each successive year following prescribed fire supports the view that nectar resources, rather than change in vegetation structure, is responsible for the observed patterns of butterfly abundance.

The limited data available for moths (macrolepidoptera and larger microlepidoptera) suggest a more complex response than that of butterflies. Total captures were 1.6 to 2.2 times greater in restored treatments in August and October compared to respective controls, similar to results for butterflies. However, in April the pattern was essentially reversed with 68-317% more captures in controls than in restored treatments. The pattern for June was somewhat transitional. Capture numbers were lowest in both April and June during the year of the prescribed fire. It appears that the relationship between moths and pine-bluestem restoration is more complex than for butterflies. Hopefully, data from additional years will clarify this relationship.

Implementation of shortleaf pine-bluestem restoration on a landscape scale in the Ouachita Mountains has restored a portion of the landscape to an ecological state more closely resembling that present prior to the influence of Europeans. Restoration also provides the habitat necessary to recover populations of the red-cockaded woodpecker while benefiting many other species that are of regional conservation concern due, at least in part, to habitat changes resulting from alteration of the pre-European fire regime. The overall success of shortleaf pine-bluestem restoration on the ONF suggests that restoration could be substantially expanded to include much of the originally fire-maintained pine communities that existed prior to European influences. Published findings and our preliminary data suggest that this restoration should increase regional nectar plant and faunal abundance and natural patterns of diversity, in addition to benefiting a number of species of conservation concern.

While this research was conducted within upland shortleaf pine habitat of the Interior Highlands, we suspect our findings should be largely applicable to much of the public holdings throughout the southeastern United States where red-cockaded woodpecker management consists of reduction of overstory basal area, reduction of the midstory, and restoration of an appropriate site-specific fire return interval.

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LITERATURE CITED

Bowman, J. L., D. R. Wood, F. J. Vilella, B. D. Leopold, L. W. Burger, Jr., and K. D. Godwin. 1999. Effects of red-cockaded woodpecker management on vegetative composition and structure and subsequent impacts on game species. Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies 53:220-234.

Brennan, L. A., J. L. Cooper, K. E. Lucas, B. D. Leopold, and G. A. Hurst. 1995. Assessing the influence of red-cockaded woodpecker colony site management on non-target forest vertebrates in loblolly pine forests of Mississippi: study design and preliminary results. Pages 309-319 *in* D. L. Kulhavy, R. G. Hooper, and R. Costa, eds. Red-cockaded woodpecker: recovery, ecology and management. Center for Applied Studies in Forestry, College of Forestry, Stephen F. Austin State Univ., Nacogdoches, Tx.

Bukenhofer, G. A., and L. D. Hedrick. 1997. Shortleaf pine/bluestem grass ecosystem renewal in the Ouachita Mountains. Trans. North Am. Wildl. Nat. Resour. Conf. 62:509-515.

Bukenhofer, G. A., J. C. Neal, and W. G. Montague. 1994. Renewal and recovery: shortleaf pine/bluestem grass ecosystem and red-cockaded woodpeckers. Proc. Ark. Acad. Sci. 48:243-245.

Burger, L. W., Jr., C. Hardy, and J. Bein. 1998. Effects of prescribed fire and midstory removal on breeding bird communities in mixed pine-hardwood ecosystems of southern Mississippi. Tall Timbers Fire Ecol. Conf. 20:107-113

Conner, R. N., D. C. Rudolph, and J. R. Walters. 2001a. The red-cockaded woodpecker: surviving in a fire-maintained ecosystem. Univ. Texas Press, Austin. 363pp.

Cram, D. S., R. E. Masters, F. S. Guthery, D. M. Engle, and W. G. Montague. 2002. Northern bobwhite population and habitat response to pine-grassland restoration. J. Wildl. Manage. 66:1031-1039.

Dana, R. P. 1991. Conservation management of the prairie skippers *Hesperia dacotae* and *Hesperia ottoe*: basic biology and threat of mortality during prescribed burning in spring. Minn. Agric. Exp. Stn. Bull. 594, Univ. Minnesota, St. Paul. 63pp.

du Pratz, L. P. 1975. The history of Louisiana. (J. G. Tregle, Jr., ed. Reprint of 1774 edition.) Louisiana State Univ., Baton Rouge. 405pp.

Featherstonhaugh, G. W. 1844. Excursion through the slave states, from Washington on the Potomac to the frontier of Mexico; with sketches of popular manners and geological notices. Murray, London. 168pp.

Fenwood, J. D., D. F. Urbston, and R. F. Harlow. 1984. Determining deer habitat capability in Ouachita National Forest pine stands. Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies 38:13-22.

Fitzgerald, J. A., and D. N. Pashley. 2000. Partners in Flight Bird Conservation Plan for the Ozark/Ouachitas (Physiographic Area 19). Version 1.0. American Bird Conservancy. (http://www.blm.gov/wildlife/plan/pl_19_10.pdf)

Foti, T. L., and S. M. Glenn. 1991. The Ouachita Mountain landscape at the time of settlement. Pages 49-66 *in* D. Henderson and L. D. Hedrick, eds. Restoration of old growth forests in the Interior Highlands of Arkansas and Oklahoma. Ouachita National Forest and Winrock International Institute, Morrilton, Ar.

Gall, L. F. 1985. Measuring the size of lepidopteran populations. J. Res. Lepid. 24:97-116.

Guldin, J. M., J. B. Baker, and M. G. Shelton. 1994. Midstory and overstory plants in mature pine/hardwood stands of the Ouachita/Ozark National Forests. Pages 29-49 *in* J. B. Baker, comp. Proc. of the symposium on ecosystem management research in the Ouachita Mountains: pretreatment conditions and preliminary findings. U.S. For. Serv. Gen. Tech. Rep. SO-112.

Haines, T. K., R. L. Busby, and D. A. Cleaves. 2001. Prescribed burning in the South: trends, purpose, and barriers. South. J. Appl. For. 25:149-153.

Jackson, J. A. 1988. The southeastern pine forest ecosystem and its birds: past, present, and future. Pages 119-159 *in* J. A. Jackson, ed. Bird conservation 3. Univ. Wisconsin Press for The International Council for Bird Preservation, United States Section, Madison.

Jansma, J., and H. H. Jansma. 1991. George Englemann in Arkansas Territory. Ark. Hist. Q. 50:225-248.

Kwilosz, J. R., and R. L. Knutson. 1999. Prescribed fire management of Karner blue butterfly habitat at Indiana Dunes National Lakeshore. Nat. Areas J. 19:98-108

Masters, R. E. 1991. Effects of timber harvest and prescribed fire on wildlife habitat and use in the Ouachita Mountains of eastern Oklahoma. Ph.D. Thesis, Oklahoma State Univ., Stillwater. 351pp.

Masters, R. E., R. L. Lochmiller, S. T. McMurry, and G. A. Bukenhofer. 1998. Small mammal response to pine-grassland restoration for red-cockaded woodpeckers. Wildl. Soc. Bull. 26:148-158.

Masters, R. E., J. E. Skeen, and J. Whitehead. 1995. Preliminary fire history of McCurtain County Wilderness Area and implications for red-cockaded woodpecker management. Pages 290-302 *in* D. L. Kulhavy, R. G. Hooper, and R. Costa, eds. Red-cockaded woodpecker: recovery, ecology and management. Center for Applied Studies in Forestry, College of Forestry, Stephen F. Austin State University, Nacogdoches, Tx.

Masters, R. E., C. W. Wilson, G. A. Bukenhofer, and M. E. Payton. 1996. Effects of pine-grassland restoration for red-cockaded woodpeckers on white-tailed deer forage production. Wildl. Soc. Bull. 24:77-84.

Masters, R. E., C. W. Wilson, D. S. Cram, G. A. Bukenhofer, and R. L. Lochmiller. 2002. Influence of ecosystem restoration for red-cockaded woodpeckers on breeding bird and small mammal communities. Pages 73-90 *in* W. M. Ford, K. R. Russell, and C. E. Moorman, eds. The role of fire in nongame wildlife management and community restoration: traditional uses and new directions. U.S. For. Serv. Gen. Tech. Rep. NE-288.

Neal, J. C., and W. G. Montague. 1991. Past and present distribution of the red-cockaded woodpecker *Picoides borealis* and its habitat in the Ouachita Mountains, Arkansas. Proc.

Ark. Acad. Sci. 45:71-75.
Nuttall, T. 1980. A journal of travels into the Arkansas Territory during the year 1819. (S. Lottinville, ed. Reprint of 1821 edition) Univ. Oklahoma Press, Norman. 361pp.
Plentovich, S., J. W. Tucker, Jr., N. R. Holler, and G. E. Hill. 1998. Enhancing Bachman's sparrow habitat via management of red-cockaded woodpeckers. J. Wildl. Manage. 62:347-354.
Pollard, E. 1977. A method for assessing changes in the abundance of butterflies. Biol. Conserv. 12:115-134.
Provencher, L., N. M. Gobris, L. A. Brennan, D.R. Gordon, and J. L. Hardesty. 2002a. Breeding bird response to midstory hardwood reduction in Florida sandhill longleaf pine forests. J. Wildl. Manage. 66:641-661.
Rudolph, D. C., and R. N. Conner. 1991. Cavity tree selection by red-cockaded woodpeckers in relation to tree age. Wilson Bull. 103:458-467.
Rudolph, D. C., R. N. Conner, R. R. Schaefer, D. Saenz, D. K. Carrie, N. R. Carrie, R. W. Maxey, W. G. Montague, J. Neal, K. Moore, and J. Skeen. 2004a. Red-cockaded woodpecker status and management: West Gulf Coastal Plain and Interior Highlands *in* R. Costa and S. J. Daniels, eds. Red-cockaded woodpecker: road to recovery. Hancock House Publishers, Blaine, Wa.
Rudolph, D. C., and C. A. Ely. 2000. The influence of fire on lepidopteran abundance and community structure in forested habitats of eastern Texas. Tx. J. Sci. 52:127-138.
SAS Institute Inc. 1988. SAS/STAT user's guide, release 6.03 edition. SAS Institute Inc., Cary, N.C.
Shannon, C. E., and W. Weaver. 1998. The mathematical theory of communication. Univ. Illinois Press, Urbana. 125pp.
Skiles, A. 1981. Arkansas climate atlas. Arkansas Energy Office, Little Rock, Ar. 93 pp.
Smith, K. G., and J. C. Neal. 1991. Pre-settlement birds and mammals of the Interior Highlands. Pages 77-104 *in* D. Henderson and L. D. Hedrick, eds. Restoration of old growth forests in the Interior Highlands of Arkansas and Oklahoma. Ouachita National Forest and Winrock International Institute, Morrilton, Ar.
Sparks, J. C. 1996. Growing-season and dormant-season fire behavior and effects on vegetation in the Ouachita Mountains, Arkansas. M.S. Thesis, Oklahoma State Univ., Stillwater. 186pp.
Sparks, J. C., R. E. Masters, D. M. Engle, and G. A. Bukenhofer. 2002. Season of burn influences fire behavior and fuel consumption in restored shortleaf pine-grassland communities. Restor. Ecol. 10: In Press.
Sparks, J. C., R. E. Masters, D. M. Engle, M. W. Palmer, and G. A. Bukenhofer. 1998. Effects of late growing-season and late dormant-season prescribed fire on herbaceous vegetation in restored pine-grassland communities. J. Veg. Sci. 9:133-142.
Sparks, J. C., R. E. Masters, D. M. Engle, M. E. Payton, and G. A. Bukenhofer. 1999. Influence of fire season and fire behavior on woody plants in red-cockaded woodpecker clusters. Wildl. Soc. Bull. 27:124-133.
Swengel, A. B. 1996. Effects of fire and hay management on abundance of prairie butterflies. Biol. Conserv. 76:73-85.

Swengel, A. B. 1998. Effects of management on butterfly abundance in tallgrass prairie and pine barrens. Biol. Conserv. 83:77-89.

Thill, R. E., P. A. Tappe, and N. E. Koerth. 1994. Wildlife habitat conditions in mature pine hardwood stands in the Ouachita/Ozark National Forests. Pages 126-143 *in* J. B. Baker, comp. Proceedings of the symposium on ecosystem management research in the Ouachita Mountains: pretreatment conditions and preliminary findings. U.S. For. Serv. Gen. Tech. Rep. SO-112.

U.S. Forest Service. 1990a. Amended land and resource management plan: Ouachita National Forest. U.S. For. Serv., Southern Region, Atlanta, Ga.

U.S. Forest Service. 1996a. Record of decision: Final environmental impact statement. Amendment 22: Renewal of the shortleaf pine/bluestem grass ecosystem and recovery of the red-cockaded woodpecker. Ouachita National Forest Land and Resource Management Plan, Scott and Polk Counties, Arkansas. U.S. For. Serv., Southern Region, Atlanta, Ga.

U.S. Forest Service. 1996b. Renewal of the shortleaf pine/bluestem grass ecosystem and recovery of the red-cockaded woodpecker: Final environmental impact statement for an amendment to the Land and Resource Management Plan, Amendment 22, Ouachita National Forest. U.S. For. Serv., Southern Region, Atlanta, Ga.

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

Williams, E. H. 1995. Fire-burned habitat and reintroductions of the butterfly *Euphydryas gilletti* (Nymphalidae). J. Lepid. Soc. 49:184-191.

Wilson, C. W. 1994. Breeding bird community composition and habitat associations in pine-oak forest and pine-bluestem woodlands in the Ouachita Mountains, Arkansas. M.S. Thesis, Oklahoma State Univ., Stillwater. 101pp.

Wilson, C. W., R. E. Masters, and G. A. Bukenhofer. 1995. Breeding bird response to pine-grassland community restoration for red-cockaded woodpeckers. J. Wildl. Manage. 59:56-67