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Amphibian and reptile responses to thinning and prescribed burning in mixed pine-hardwood forests of northwestern Alabama, USA

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ABSTRACT

We evaluated the response of amphibians and reptiles to two levels of prescribed burning and three levels of thinning using a field experiment consisting of a before–after, control-impact, and factorial complete block design over a four year period in the William B. Bankhead National Forest located in northwestern Alabama. We captured 2643 individuals representing 47 species (20 amphibians and 27 reptiles) during 3132 trap nights. Pre-treatment captures varied widely for both amphibians and reptiles among the stands designated for management, which was likely due to forest structural changes caused by tree mortality resulting from Southern Pine Beetle (*Dendroctonus frontalis*) infestations. Within each amphibian and reptile species assemblage, we observed species-specific associations with specific treatments and environmental characteristics. In regards to individual species responses, Eastern Fence Lizards (*Sceloporus undulatus*) increased in thin-with-burn treatments and Green Anoles (*Anolis carolinensis*) tended to increase in all thinned stands. North American Racers (*Coluber constrictor*) increased in thin-only plots primarily during the second post-treatment year. Mississippi Slimy Salamander (*Plethodon mississippi*) captures tended to decrease in all treatment stands throughout the study period, which may been due to either drier environmental conditions during post-treatment sampling or natural population cycling. Pool-breeding amphibian captures were more likely related to the hydroperiod of aquatic breeding environments within 290 m of survey locations rather than forest treatments. Our results illustrate that forest restoration through tree thinning can positively influence certain reptile species with limited impacts on amphibians in upland, pine-dominated forests of northern Alabama. However, as our forest stands are scheduled to be burned every 3–5 years, continued monitoring is necessary to understand the impacts of repeated disturbances.

1. Introduction

Ecological disturbances are events that disrupt ecosystem, community, or population structure and change resource and substrate availability in the physical environment (White and Pickett, 1985). Understanding the relationship between disturbances and animal responses is essential for longterm species conservation, and factors such as spatial and temporal scale, frequency, and intensity all play important roles in determining species responses to disturbance (Petraitis, 1989). Anthropogenic disturbances (e.g., fragmentation, forest conversion, disturbance suppression) not consistent with an ecosystems disturbance history may have long-lasting negative impacts on species conservation because they alter habitat conditions in a manner inconsistent with the environmental conditions of which particular species has evolved (Turner et al., 1989). In areas where the historical patterns of disturbance have been disrupted by anthropogenic means, management practices such as burning and tree removal can be used as a surrogate for stochastic disturbance events to increase habitat connectivity and maintain or restore focal habitats (Drever et al., 2006).

There has been heightened interest in the response of amphibians and reptiles to disturbances, including forest management (Hawkes and Gregory, 2012; Russell et al., 2004; Semlitsch et al., 2009; Steen et al., 2010), which is most likely due to the importance of herpetofauna in ecological food webs (Fitch, 1949; Burton and Likens, 1975; Wyman, 1998) and the apparent worldwide declines of these species (Gibbons et al., 2000; Stuart et al., 2004). Due to the range of impacts from forest management, it is important to consider the type of management strategy when evaluating amphibian and reptile responses. With respect to amphibians, clearcut harvesting has been shown to have the greatest negative impacts to terrestrial plethodontid salamanders (e.g., Homyack

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and Haas, 2009; Karraker and Welsh, 2006; Knapp et al., 2003; Perkins and Hunter, 2006) due to alteration of environmental characteristics that increase the risk of dessication. Pool-breeding amphibians that have evolved in forested ecosystems with a relatively long time between disturbances are also particularly sensitive to the environmental changes caused by clearcutting forest management (Fredenfields et al., 2011; Patrick et al., 2006; Semlitsch et al., 2009). The effects of even-age shelterwood harvesting (Harpole and Haas, 1999; Knapp et al., 2003) and thinning operations (Naughton et al., 2000; Grialou et al., 2000) on plethodontid salamanders appear to be species-specific, whereas uneven-aged management, such as group and single tree selection, either has minimal (Messere and Ducey, 1998; McKenny et al., 2006) or negative effects (Cromer et al., 2002; MacCracken, 2005). Although fewer studies have evaluated reptile response to forest management practices compared to amphibians in the eastern and southeastern United States (Greenberg, 2001), reptiles tend to exhibit species-specific responses to most tree harvesting operations (e.g., Adams et al., 1996; Greenberg and Waldrop, 2008; Renken et al., 2004; Steen et al., in press a) and generally benefit from management and disturbance conditions that mirror the ancestral habitat conditions in which a given species has evolved (Steen et al., 2010, in press b).

Studies examining herpetofaunal responses to prescribed burning in the eastern US have generally focused on certain ecoregions (e.g., southeastern Coastal Plain; e.g., Bishop and Haas, 2005; Schurbon and Fauth, 2003; Means et al., 2004) or species groups (e.g., amphibians, Ford et al., 2010; Russell et al., 2004). Overall, prescribed fire appears to have negative short-term effects on amphibians that inhabit ecosystems that are not fire prone (Cole et al., 1997; Kirkland et al., 1996; Mcleod and Gates, 1998) or forest stands where the natural fire regime has been suppressed (Schurbon and Fauth, 2003; Means et al., 2004). Other studies have found that fire has negligible impacts (Ford et al., 1999; Moseley et al., 2003; Greenberg and Waldrop, 2008) or short-term positive effects (Mushinsky, 1985; Wilgers and Horne, 2006) on certain amphibian species. Prescribed burning appears to have either positive impacts for some reptile species (Moseley et al., 2003; Wilgers and Horne, 2006; Greenberg and Waldrop, 2008; Perry et al., 2012) or no measurable impacts (Mcleod and Gates, 1998). In longleaf pine ecosystems of the southeast, prescribed burning is essential to maintain habitat for herpetofauna native to these ecosystems (Russell et al.,1999; Means et al., 2004; Yager et al., 2007). The disparity of amphibian and reptile species responses to fire disturbance is primarily related to whether a species has evolved in an ecosystem that experiences periodic disturbances through fire (Pilliod et al., 2003; Means et al., 2004; Steen et al., 2010). However, our knowledge of herpetofaunal responses to fire throughout most areas of the United States remains insufficient (Bury, 2004) and inconsistencies among studies including fire characteristics (e.g., intensity, fire return interval), specific taxa examined, and study design make it difficult to compare management implications among studies (Russell et al., 1999).

Most studies of herpetofaunal responses to forest management have taken a retrospective approach without data from pre-harvest conditions (Russell et al., 2004). The lack of pre-treatment data limits study inference and effective evaluation of forest management practices for ecosystem restoration and wildlife habitat management. In addition, few studies have evaluated the impacts of simultaneous thinning and prescribed burning forest management on amphibians and reptiles. In this study, we took a large-scale, replicated, stand-level approach to evaluate herpetofaunal responses to forest management practices (thinning and prescribed burning) as a part of a larger study evaluating ecosystem response to forest restoration. Because forest thinning results in canopy cover reduction and increased air and soil temperatures, we expected to see positive changes (i.e., increases in abundance) for many reptile species, specifically in thinning treatments. Additionally, we expected that amphibian responses would decline greatest in thin with burn treatments due to the simultaneous reduction of canopy cover and the forest litter layer. Overall, we predicted that the impacts of forest restoration on amphibians and reptiles would be related to the behavioral and physiological adaptations (e.g., thermoregulation, moisture requirements) that particular species have acquired in relation to the disturbance history of the southern Cumberland Plateau.

2. Materials and study area

2.1. Study area

We conducted this study in the northern portion of the William B. Bankhead National Forest (BNF), located in Lawrence, Winston, and Franklin Counties of northwestern Alabama (Fig. 1). The BNF is a 72,800 ha multi-use forest located along the highly dissected portion of the southern Cumberland Plateau (Smalley, 1982; Gaines and Creed, 2003). Soils within this region are typically composed of Hartsells-Rock and limestone-Hector (Smalley, 1982). Mixed forests of the southern Cumberland Plateau tend to be dominated by oak-hickory forest types (McWilliams, 1991) except in areas where pines were planted for commercial purposes. Loblolly Pine (*Pinus taeda*), which is a native tree species in the southeastern United States, was used to re-establish forest conditions in abandoned agricultural and heavily timbered areas during the 1930s (Gaines and Creed, 2003). Reforestation efforts along with natural growth have resulted in 31,600 ha of *P. taeda* throughout the BNF (Gaines and Creed, 2003). For the past decade, Southern Pine Beetle (*Dendroctonus frontalis*) infestations have affected *P. taeda* stands, producing large numbers of standing dead trees and increased fuel loads, elevating the risk of damaging wildfires. In 2003, the BNF initiated a forest restoration plan (FRP) to reduce wildfire risk and promote growth of natural upland, hardwood forest communities through tree thinning and prescribed fire (Gaines and Creed, 2003). The BNF's FRP mirrors regulations in the Healthy Forest Restoration Act, which authorizes advanced vegetation management projects when specified conditions (e.g., existence of insect or disease epidemic) pose a threat to ecosystem health (Healthy Forest Restoration Act, 2003). In 2004 the BNF began a partnership with Alabama A&M University to evaluate the impacts of restoration strategies (i.e., thinning and burning) on the overall forest ecosystem through a multi-disciplinary (i.e., wildlife, soils, vegetation, molecular studies, and human dimensions) approach.

The forest stands examined in this study were located on upland sites composed of loblolly pine 25–50 years of age that also possessed a hardwood component (Gaines and Creed, 2003; Schweitzer and Tadesse, 2004). At the time of this study, these stands had not been harvested for approximately 30 years and each stand had varying levels of damage from *D. frontalis* (Gaines and Creed, 2003).

2.2. Experimental design and forest treatments

Our experiment consisted of a before-after, control-impact (BACI), complete block design with a 2×3 factorial arrangement of three thinning levels (no thin, 11 $m^2\,ha^{-1}$ residual basal area [BA], and 17 $m^2\,ha^{-1}$ residual BA) and two burn levels (no burn and burn), which resulted in six treatments that included a control [no thin and no prescribed burn], burn [prescribed burn and no thin], light thin [17 $m^2\,ha^{-1}$ BA retention and no burn], heavy thin [11 $m^2\,ha^{-1}$ BA retention and no burn], light thin with burn [17 $m^2\,ha^{-1}$ BA retention with prescribed burn], and heavy thin

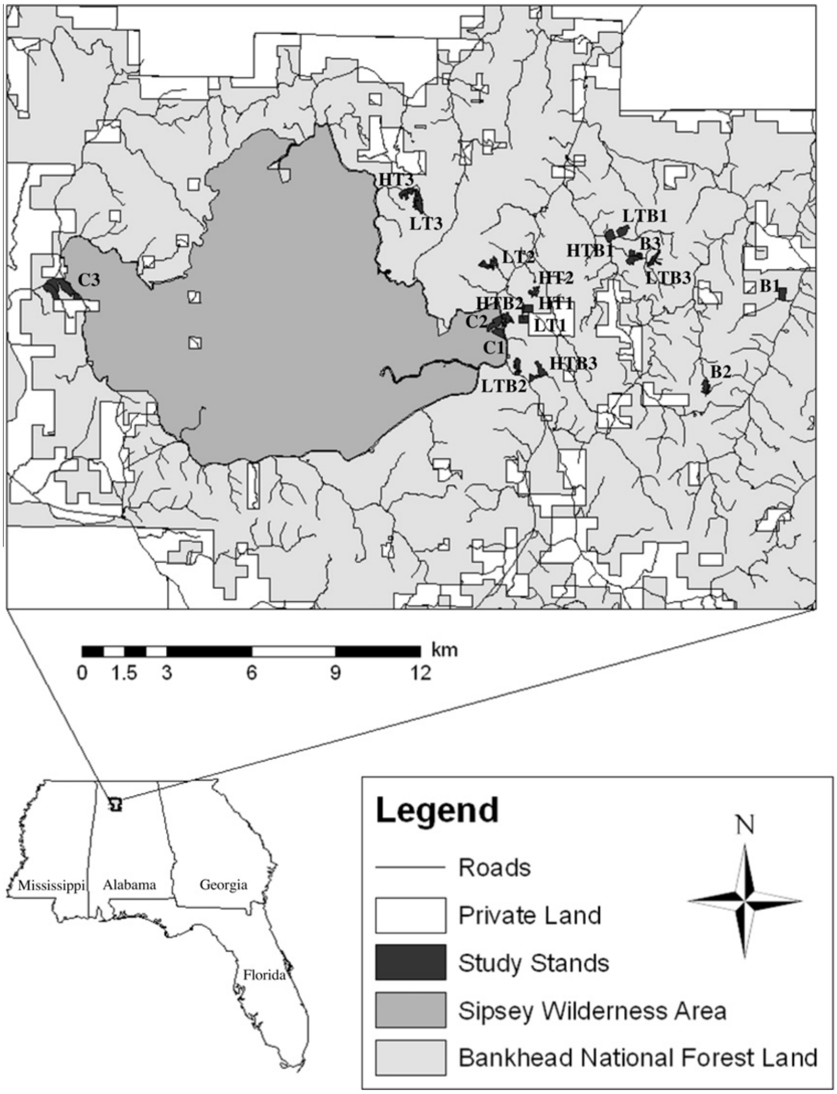

Fig. 1. Locations of study stands within the northwestern portion of the William B. Bankhead National Forest, Alabama, USA. Forest treatments consisted of: C – control, B – burn, LT – light thin, HT – heavy thin, LTB – light thin with burn, and HTB – heavy thin with burn. Numbers after treatment abbreviations correspond to treatments within a block.

with burn [11 $m^2\,ha^{-1}$ BA retention with prescribed burn]. Each treatment (~9 ha in size) was replicated three times across the landscape (Fig. 1), equaling 18 total treatments (i.e., stands). The process of allocating a particular management strategy to a stand was not fully randomized because treatment designations had to align with the longterm management goals of the BNF. For example, forest stands assigned to prescribed burn treatments had to be located in a portion of the BNF that was designated a burn area in the original FRP. Distance between treatments varied greatly, with some treatments adjacent to one another and other treatments much further apart (maximum – 26 km). In addition, distance among treatment replicates averaged 5.3 km (range 3.6–10.8 km). Due to difficulties implementing this large-scale study in a single year, treatments were blocked temporally (i.e., year). Block one was treated during the summer of 2005, and blocks two and block three were treated during the summer and fall of 2006. All harvesting was thin-from-below, which targets trees of suppressed and intermediate crown classes (i.e., generally the tree classes that receive little to no direct sunlight) to provide limited resources (e.g., water) to dominant and co-dominant trees (Smith et al., 1997). Harvesting was completed by feller bunchers and trees were harvested until the desired residual BA was achieved. Hardwood tree species, such as *Quercus spp.* and *Carya spp.*, were preferentially retained during harvesting.

Prescribed burns consisted of dormant season fires (February–March), which were ignited when air temperatures were low and relative humidity was high. Backing fires were initiated to ensure that prescribed burns were limited to understory and litter layers. Prescribed burns were ignited with drip torches and diminished naturally when the available fuel was consumed.

2.3. Amphibian and reptile sampling

The primary focus of this study was to determine amphibian and reptile responses to thinning and prescribed burning management. To do this, we developed a trapping system consisting of three drift fences (aluminum flashing) 15 m in length radiating 120° from a central triangular box trap (Sutton et al., 2010). We chose this design because large box traps are successful for capturing medium- and large-bodied snake species (Burgdorf et al., 2005) in addition to other amphibian and reptile species. To determine the location of a drift-fence array within a stand, we divided each

Table 1
Environmental, treatment, and treatment year covariates used for multivariate analyses. Unbolded variables correspond with microhabitat and microclimate data collected within each treatment stand in the William B. Bankhead National Forest, Alabama, USA.

Habitat variable	Code	Description
Percent litter[a,b]	% litter	Presence (%) of ground cover such as leaves or small woody debris measured at every 0.5 m along the habitat transect
Percent bare ground[a]	% bare	Absence (%) of ground cover (e.g., exposed soil) measured at every 0.5 m along the habitat transect
Percent herbaceous[a]	% herb	Presence of non-woody stems (%) such as grasses, ferns, and *Smilax* and *Vitus* sp. measured at every 0.5 m along the habitat transect
Percent woody[a,b]	% wood	Presence of any woody stems (%) such as seedlings and large trees measured at every 0.5 m along the habitat transect; woody stems taller than one meter had to contact transects directly to be counted
Percent rock	% rock	Presence of rocky substrate (%) greater than 10 cm in size measured at every 0.5 m along the habitat transect
Percent CWD[a,b]	% CWD	Presence of any fallen woody debris larger than 10 cm in diameter (must touch the ground somewhere along the length to be counted) measured at every 0.5 m along the habitat transect
Percent slash[a]	% slash	Presence of any woody debris (%) composed of two or more stems 30 cm or higher from the ground (e.g., fallen treetops) measured at every 0.5 m along the habitat transect
CWD volume[a]	CWD vol	Calculated as volume of a cylinder (m^3) for each enumerated CWD
Litter depth[a,b]	L_dep	Determined by measuring depth of the substrate to the nearest 0.5 cm with a metric ruler measured at every 2 m along the habitat transect
Canopy cover[a]	Can cov	Estimated with a spherical densiometer as the sum percentage of open points subtracted from 100% measured at every 5 m along the habitat transect
Forest level 1[a]	For lev 1	Percent coverage of forest levels $\leqslant 2$ m (classified as ground cover) measured at every 5 m along the habitat transect
Forest level 2[a]	For lev 2	Percent coverage of forest levels >2–$\leqslant 4$ m (classified as understory) measured at every 5 m along the habitat transect
Forest level 3[a,b]	For lev 3	Percent coverage of forest levels >4–$\leqslant 6$ m (classified as midstory) measured at every 5 m along the habitat transect
Forest level 4[a]	For lev 4	Percent coverage of forest levels > 6 m (classified as overstory) measured at every 5 m along the habitat transect
Air temperature[a,b]	Air temp	Air temperature (°C) sampled with a datalogger at 12 h intervals starting at 2:00 PM
Soil temperature[a]	Soil temp	Soil temperature (°C) sampled with a datalogger at 12 h intervals starting at 2:00 PM
Relative humidity	Rel hum	Relative humidity (%) sampled with a datalogger at 12 h intervals starting at 2:00 PM
Light intensity[a]	Light	Light intensity (lumens/ft^2) sampled with a datalogger at 12 h intervals starting at 2:00 PM
Basal area[a]	Bas area	Cross sectional area of all trees 15 cm and larger in diameter within a 0.08 ha circular plot. Overall plot value (m^2/ha) taken as an average of five 0.08 hectare plots
Breeding pond[b]	Pond	Total number of breeding ponds located within 290 m of an amphibian and reptile sampling location
Treatment[b]	NA	Dichotomous variable designating a specific treatment. Coded as follows: C – Control, B – Burn, LT – Light Thin, HT – Heavy Thin, LTB – Light Thin with Burn, and HTB – Heavy Thin with Burn
Treatment year[b]	NA	Dichotomous variable designating a specific treatment year. Coded as follows: Pre – Pre-treatment, Post 1 – Post-treatment year 1, and Post 2 – Post-treatment year 2

[a] Corresponds with environmental covariates used in principal components analysis.

[b] Corresponds with environmental, treatment, and treatment year covariates used in canonical correspondence analysis.

stand into quadrants corresponding to the four cardinal directions and installed the drift-fence array into one of these randomly selected quadrants. The drift-fence arrays were removed before treatment implementation and were reinstalled afterward at the same locations. We collected amphibian and reptile trap data over a period of four years (i.e., 2005–2008). Due to the staggered treatment schedule of the blocks (i.e., block one implemented in July 2005 and blocks two and three implemented in May 2006), we collected pre-treatment data over a period of three months (April–June 2005) for block one and six months (May 2005–August 2005 and March 2006–May 2006) for blocks two and three. We collected two years of post-treatment data for blocks two and three, and three years of post-treatment data for block one. We excluded the third year of post-treatment data from all analyses. Pre- and post-treatment data were constrained to the same time periods to permit comparisons within each block across years.

We sampled herpetofauna intermittently throughout March and April to target days that were most likely to result in highest captures (i.e., warmer days and rainy nights) and began continuous sampling from the beginning of May until September of each year. Traps were surveyed by block(s) depending on weather conditions and manpower, with the order of blocks and treatments within a block randomly determined *a priori*. We checked traps daily between 0700 and 1400 h (CST) to minimize animal mortality due to dessication and/or solar exposure. After recording data (e.g., species, sex, snout-vent length, and mass), we marked each individual with a stand-specific mark through a single toe-clip (lizards, anurans, and salamanders), a single scale clip (snakes), or a single scute etch (turtles) to avoid counting the same individuals in subsequent captures (Enge, 1997). We released all marked individuals at a distance greater than 10 m from the side of the drift fence where they were captured.

2.4. Microclimate and microhabitat sampling

We installed one HOBO© (Onset Computer Corp.) datalogger at each drift-fence array to record air temperature, soil temperature, relative humidity, and light intensity (Table 1). We programmed dataloggers to record measurements every 12 h starting at 1400 hr to record microclimate data at the warmest and coolest periods of the day. Because the pre-treatment sampling period was short in block one (i.e., treatment implementation began in July of the same year that pre-treatment sampling began), we only included climate data from May 15 to July 15 for all treatments across all survey years as a means to standardize comparisons among climate data. We recorded pre- and post-treatment habitat features annually via line-transect surveys. We determined plot placement *a priori* via a random compass bearing (0–360°) and distance (30–50 m) originating from the center of the drift-fence trapping arrays. We restricted habitat surveys to occur within 30–50 m of the trapping arrays to avoid habitat disturbances created during trap installation, but to also maintain relevancy between the herpetofaunal capture and microhabitat data. We sampled three total habitat plots in each stand, with each sampling location composed of four 10 m belt transects extending outwards in the four cardinal directions from the randomly determined habitat plot center. We used a 2 m piece of 1.9 cm diameter polyvinyl chloride pipe centered on the transect line and recorded the presence or absence of a suite of microhabitat variables across the transect (Table 1). To estimate percent cover of litter, herbaceous and woody vegetation, rock, CWD, and slash, we recorded presence and absence of these attributes every 0.5 m along the transect and summed the total number of presences (80 total survey locations per habitat plot) individually for each habitat variable, divided this number by the total amount of possible survey locations, and multiplied by 100

Table 2
Herpetofaunal species assemblages included in Canonical Correspondence Analysis. Only species with >5 captures were included in these analyses. Species accounts found in Jensen et al. (2008), Mount (1975), and Niemiller and Reynolds (2011) were used to determine species assemblages.

Assemblage	Species	Support
Southeastern lizards	*Anolis carolinensis* *Plestiodon fasciatus* *Plestiodon lateralis* *Sceloporus undulatus* *Scincella lateralis*	Includes lizard species that are typically encountered in mixed pine-hardwood forested ecosystems of northern Alabama
Upland, large-bodied snakes	*Agkistrodon contortrix* *Coluber constrictor* *Crotalus horridus* *Heterodon platirhinos* *Lampropeltis nigra* *Pantherophis guttatus* *Pantherophis spiloides*	Includes snake species with a maximum snout-vent length > 900 mm typical of mixed pine-hardwood forested ecosystems in northern Alabama
Lungless, terrestrial salamanders	*Plethodon ventralis* *Plethodon mississippi* *Pseudotriton r. ruber*	Includes terrestrial, lungless salamanders within the family Plethodontidae. Some species within this group spend a majority, if not all of their life in the terrestrial environment.
Ephemeral, pool-breeding amphibians	*Ambystoma maculatum* *Ambystoma opacum* *Gastrophryne carolinensis* *Hyla chrysoscelis* *Hyla cinerea* *Pseudacris brachyphona* *Scaphiopus holbrookii*	Includes amphibian species that primarily use ephemeral, fishless water sources as breeding locations
Semi-permanent, pool-breeding amphibians	*Acris crepitans* *Anaxyrus fowleri* *Lithobates catesbeianus* *Lithobates clamitans* *Lithobates palustris* *Lithobates sphenocephalus*	Includes amphibian species that use semi-permanent and permanent water sources as breeding locations

(e.g., 68 litter “presences”/80 potential survey sites $*$ 100 = 85% litter cover). We derived an overall stand estimate for each habitat variable by averaging the values from each of the three habitat plots.

We estimated percent canopy cover and vertical forest structure every 5 m along the habitat transect. We determined percent forest structure cover on a scale of 1–4 (Table 1) according to the designations described in Forest Inventory and Analysis (1998) and summed the number of occurrences for each level within each belt transect, divided this value by the number of sampling sites, and multiplied by 100 (e.g., 7 forest level 2 “presences”/8 potential survey sites $*$ 100 = 88% forest level 2 cover). We used a spherical densiometer to obtain a canopy cover estimate at each 5 m interval and as with the other variables derived an overall stand estimate by averaging the values from each of the three habitat plots. We measured litter depth every 2 m along the transect and estimated the volume of CWD > 10 cm diameter that were intersected by the transect (Table 1). We repeated habitat surveys at the same locations during pre- and post-treatment years.

2.5. Data analysis

2.5.1. Species comparisons

To gain a better perspective on species responses to disturbance, it is important to evaluate whether species responses were due to true changes or unequal detection among treatment. We estimated detection probabilities for each amphibian or reptile species with greater than 100 unique captures using program PRESENCE (v.3.0; Hines, 2010). We used a single-season modeling approach with each year treated as a single sampling event (MacKenzie et al., 2002). This resulted in three total sampling events (i.e., pre-treatment, post-treatment year one, and post-treatment year two). Because we were only interested in evaluating factors affecting detection, we kept occupancy constant across models. We evaluated two potential models for each species and species group, including a null model (no covariates and assuming a constant detection probability) and a model that included treatment covariates coded to represent the six treatments. To assess the fit of the resulting models, we calculated an over-dispersion parameter ($\hat{c}$) and used this value to adjust the fit of the resulting models for each species or species group (Mackenzie et al., 2006).

We evaluated the impacts of forest management treatments on individual species that showed constant detections among forest treatments. We excluded all recaptures from this analysis, and divided the total number of captured individuals by the total number of trap nights (one trap night = one trap opened for 24 h) to correct for differences in trapping effort among years. We multiplied this corrected capture number by 100 to estimate the number of animals captured per 100 trap nights (Greenberg and Waldrop, 2008). We used logistic, log-normal, and square root transformations to approximate a normal distribution for the amphibian and reptile capture data prior to analysis. We used mixed models (PROC MIXED) analysis of variance (ANOVA; SAS v. 9.1.3) to test changes in herpetofaunal captures between pre- and post-treatment samples (within subject factor), among the treatments (between subject factor), and their interactions. Mixed models permit the analysis of random effects (i.e., block) along with fixed effects (i.e., treatment), while accounting for repeated samples (i.e., year). We declared significance at an alpha level $\leqslant 0.05$ for all statistical analyses and used a Tukey test for post hoc comparisons. We reported means (±SE), unless otherwise indicated.

2.5.2. Habitat comparisons

We used principal component analysis (PCA) to examine relationships among habitat and climate parameters using SPSS v. 15.0. We also used this test to reduce dimensions for subsequent analyses, which helped to limit Type 1 statistical errors associated with testing the change in each habitat variable singularly. We used logistic, log-normal, square root, and $1/x$ transformations to approximate a normal distribution for habitat data prior to analysis. We excluded percent humidity and percent rock cover from the PCA due to our inability to approximate normal distributions for

Table 3
Significance levels from mixed model analysis of variance for basal area, habitat, and amphibian and reptile variables for 1 year before and 2 year post-treatment (2005–2008) in the William B. Bankhead National Forest, Alabama, USA. Bolded headings represent comparisons were treatment and year interactions were detected.

	B	T	B ∗ T	Yr	B ∗ Yr	T ∗ Yr	B ∗ T ∗ Yr[b]	Pairwise comparisons[c]
Habitat								
Basal Area	0.45	<0.0001	0.75	<0.0001	0.99	<0.0001	0.79	Treat ∗ Yr: (2) C B LT LTB HTB HT; (3) C B LT LTB HTB HT
PC 1[a]	<0.01	<0.0001	0.21	<0.0001	<0.01	<0.0001	0.44	Treat ∗ Yr: (2) C B LT LTB HT HTB; (3) C B LT LTB HT HTB
PC 2[a]	0.01	0.01	0.39	<0.01	0.33	0.74	0.89	B: B NB T: NT LT HT
PC 3[a]	0.39	0.70	0.07	0.19	0.69	0.99	0.82	NS
Species responses								
Sceloporus undulatus	0.21	<0.001	0.69	<0.01	0.01	0.50	0.95	Treat ∗ Yr: (3) C B HT LT LTB HTB
Anolis carolinensis	0.06	<0.0001	0.06	<0.01	0.93	0.43	0.75	T: (2) B C LT HT LTB HTB
Plestiodon fasciatus	0.99	0.17	0.16	0.90	0.44	0.05	0.25	Treat ∗ Yr: (2) B LT C LTB HTB HT
Scincella lateralis	0.07	0.89	0.15	0.03	0.79	0.47	0.72	Yr: 2 3 1
Coluber constrictor	0.13	0.14	0.07	<0.01	0.89	0.02	0.78	Yr: 1 2 3
Agkistrodon contortrix	0.28	0.78	0.71	0.82	0.79	0.70	0.68	NS
Plethodon glutinosus	0.73	0.12	0.10	<0.0001	0.57	0.91	0.23	Yr: 1 2 3
Anaxyrus fowleri	0.26	0.88	0.15	0.33	0.99	0.42	0.93	NS

[a] Represents biologically relevant principal components.
[b] Model terms as follows: B (Burn), T (Thin), and Yr (Year).
[c] Underlines that do not overlap represent significant differences. Abbreviations as follows: C (Control), B (Burn), LT (Light Thin), HT (Heavy Thin), LTB (Light Thin with Burn), HTB (Heavy Thin with Burn), T (Thin), B (Burn), NB (No Burn), NT (No Thin), NS (Non-significant), and Treat * Yr (Treatment and Year). Numbers in parentheses represent the following year designations: (1) pre-treatment, (2) post-treatment year 1, and (3) post-treatment year 2. Tukey tests were used for all pairwise comparisons.

these data. We used mixed models ANOVA to examine changes in ecologically relevant principal components derived from pre- and post-treatment habitat data. As with the species comparisons, we declared significance at an alpha level $\leqslant 0.05$ and used a Tukey test for post hoc comparisons. We reported means (±SE), unless otherwise indicated.

2.5.3. Species and habitat comparisons

We used canonical correspondence analysis (CCA) to examine the impacts of forest management on amphibian and reptile species assemblages by evaluating the simultaneous influence of environmental, treatment, and treatment year covariates on species captures. CCA is a direct gradient analysis technique that is constrained by a set of *a priori* environmental characteristics that are hypothesized to influence species distribution patterns (Ter Braak, 1995). Prior to analysis we examined correlations and relationships between habitat variables using PCA (Sutton et al., 2010). In each CCA we included seven environmental variables previously identified as biologically important at these study sites (see Sutton 2010) and nine binary variables representing both treatment and treatment year effects (Table 1), similar to Hawkes and Gregory (2012). We used CANOCO v. 4.5 to complete five separate CCAs that corresponded with distinct herpetofaunal species assemblages, including southeastern lizards, upland large-bodied snakes, terrestrial lungless salamanders, ephemeral pool-breeding amphibians, and semi-permanent pool-breeding amphibians (Table 2). For CCAs examining ephemeral pool-breeding amphibians and semi-permanent pool-breeding amphibians, we also included a covariate that represented the number of aquatic breeding habitats > 0.01 ha located within 290 m of each trapping array. Although pool-breeding amphibians can travel much further distances in the terrestrial environment, we based this distance on the buffer size recommendations for aquatic breeding habitats as described in Semlitsch and Bodie (2003). For all CCAs, we excluded species with less than five captures over the entire study period. Ter Braak (1995) suggests that rare species have little influence on the analysis and can be removed to improve the interpretability of the CCA. We pooled trap night corrected herpetofaunal capture data across replicates to permit the assessment of environmental, treatment, and treatment year effects similar to Hawkes and Gregory (2012). We used biplot scaling to infer species relationships and downweighted the influence of rare species on the analysis. We assessed statistical significance of the resulting canonical relationships with a Monte-Carlo permutation test (9999 permutations) and created joint biplots for each species assemblage to evaluate the relationships among species data and environmental, treatment, and treatment year covariates.

3. Results

3.1. Harvesting data

To understand the extent of overstory tree management, we compared the amount of overstory basal area in each stand before and after treatment. Overstory basal area was similar among stands before treatment and ranged from 26.8 m^2/ha to 29.1 m^2/ha, but was reduced to an average of 12.7 m^2/ha in all thinned stands after treatment ($F_{2,24} = 29.0$; $p < 0.0001$; Table 2). Post hoc analyses revealed that the basal area of overstory trees did not differ statistically between heavy thin (11.3 ± 0.9 m^2/ha [heavy thin] and 10.9 ± 0.8 m^2/ha [heavy thin with burn]) and light-thin (14.6 ± 1.2 m^2/ha [light thin] and 13.8 ± 0.3 m^2/ha [light thin-with-burn]) stands after treatment (Table 3).

3.2. Trapping summary

We captured 2643 amphibians and reptiles representing 47 species during 3132 trap nights (pre-treatment [564 total trap nights], post-treatment year 1 [1086 total trap nights], post-treatment year 2 [1212 total trap nights], and post-treatment year 3 [270 total trap nights]; Appendix A). The most commonly captured lizard and snake species were Green Anoles (*Anolis carolinensis*, $n = 283$) and Copperheads (*Agkistrodon contortrix*, $n = 178$), respectively, whereas Mississippi Slimy Salamanders (*Plethodon mississippi*, $n = 674$) and Fowler's Toads (*Anaxyrus fowleri*, $n = 177$) represented the most commonly captured salamanders and anurans, respectively (Appendix A). The Woodland Box Turtle (*Terrapene c. carolina)* was the most commonly captured turtle species ($n = 8$; Appendix A). Overall, 371 individuals were recaptures, with *A. carolinensis* being the most commonly recaptured reptile species

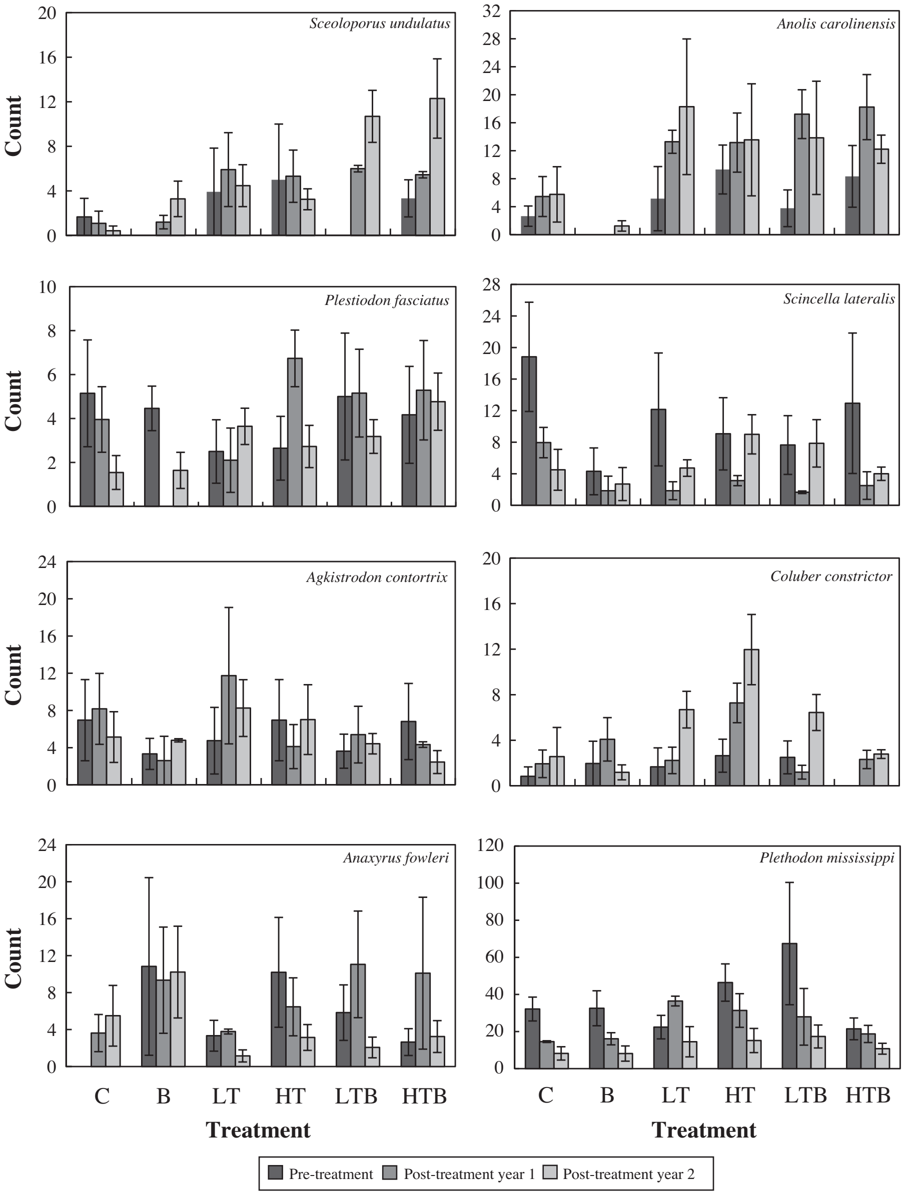

Fig. 2. Amphibian and reptile captures (mean ± SE) one year before and two years after prescribed burning and thinning treatments in the William B. Bankhead National Forest of northwestern Alabama, USA (2005–2008).

(146 recaptures) and *A. fowleri* being the most commonly recaptured amphibian species (65 recaptures; Appendix A).

3.3. Species detections

We conducted a coarse analysis of species detection probabilities via a null (constant detection) and a model that permitted detection to vary by treatment. Overall, the model with constant detection had the highest support for the individual species comparisons. Individual species detection probabilities (±SE) for these models were as follows: Eastern Fence Lizard (*Sceloporus undulatus* [0.74 ± 0.07]), *A. carolinensis* (0.84 ± 0.05), Common Five-lined Skink (*Plestiodon fasciatus* [0.81 ± 0.05]), Little Brown Skink (*Scincella lateralis* [0.83 ± 0.05]), *A. contortrix* (0.81 ± 0.05), North American Racer (*Coluber constrictor* [0.71 ± 0.07]), *A. fowleri* (0.72 ± 0.07), and *P. mississippi* (0.96 ± 0.03). Overall, these detection results provide some evidence that the probability of detection for the most commonly encountered species were similar among forest stands and that the observed changes were not due to unequal detection among treatments.

3.4. Species responses

We evaluated individual species responses to forest management for species that displayed constant detection rates throughout the study period. *S. undulatus* captures in burn-only stands differed by year ($F_{2,36} = 4.91$; $p < 0.05$), with higher captures in heavy thin with burn treatments compared to control treatments during the second post-treatment year (Table 3 and Fig. 2). *A. carolinensis* captures were greater in thin and thin with burn stands throughout all years (Fig. 2) and were significantly greater in these stands compared to prescribed burn treatments during the first post-treatment year (Table 3). *S. lateralis* captures differed by year in all treatment stands ($F_{2,36} = 3.23$; $p = 0.05$; Table 3) and initially decreased in all treated stands during the first post-treatment year (Fig. 2). *P. fasciatus* captures differed by year in thinned stands ($F_{2,34} = 2.61$; $p = 0.05$), which were significantly greater in heavy-

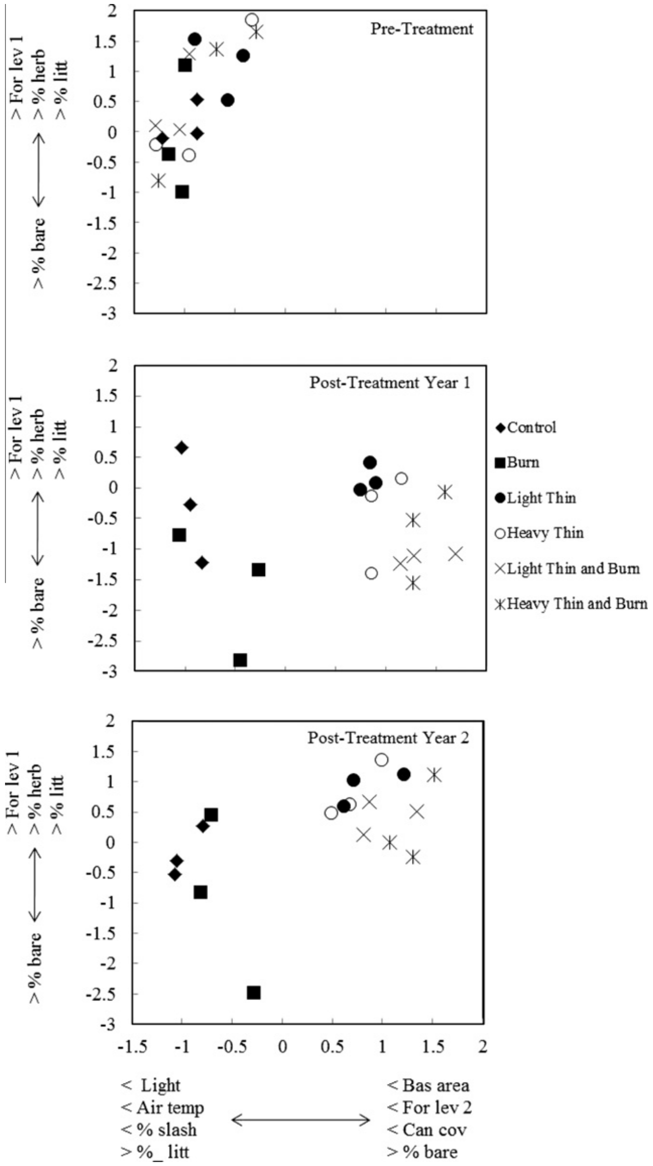

Fig. 3. Positions of managed forest stands within the space of the first two principle components based on habitat variables sampled in the William B. Bankhead National Forest, Alabama, US.A (2005–2008). From top to bottom: pre-treatment, post-treatment year one, and post-treatment year two. Component 1 (*x*-axis) represents a basal area, air temperature, and canopy cover gradient, whereas component 2 (*y*-axis) represents a groundcover gradient. See Table 1 for habitat and climate variable codes and definitions.

thin treatments compared to prescribed burn treatments during the first post-treatment year (Table 3). *C. constrictor* captures differed by year in thinned stands ($F_{4,36} = 3.45$; $p < 0.05$; Table 3), with captures greatest in heavy-thin treatments during the second post-treatment year (Fig. 2). *A. contortrix* captures were highly variable and did not appear to be impacted by forest treatments (Fig. 2). *P.mississippi* captures differed by year across all treatments (Table 3) and tended to decline across all years with a similar pattern across all treatments (Fig. 2). *A. fowleri* captures varied among treatments with no consistent pattern through the study period (Fig. 2).

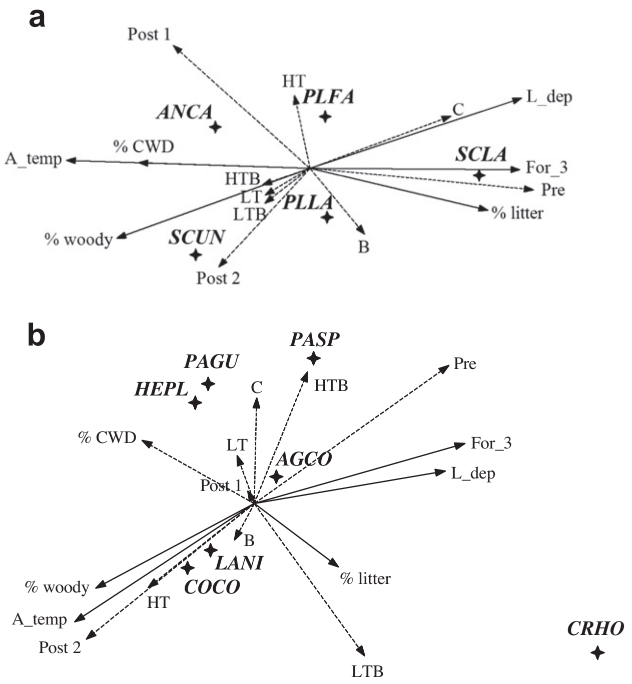

Fig. 4. Canonical correspondence analysis biplot showing relationships between the abundance of species within a southeastern lizard assemblage (a) and a large-bodied upland snake assemblage (b) and environmental, treatment, and treatment year covariates. Four-letter codes associated with the star symbol represent species responses, solid black lines represent environmental covariates, and dashed lines represent binary covariates (treatment and treatment year variables). AGCO – *Agkistrodon contortrix*, ANCA – *Anolis carolinensis*, COCO – *Coluber constrictor*, CRHO – *Crotalus horridus*, HEPL – *Heterodon platirhinos*, LANI – *Lampropeltis nigra*, PAGU – *Pantherophis guttatus*, PASP – *Pantherophis spiloides*, PLFA – *Plestiodon fasciatus*, PLLA – *Plestiodon laticeps*, SCLA – *Scincella lateralis*, SCUN – *Sceloporus undulatus*. Please refer to Table 1 for environmental, treatment, and treatment year covariate descriptions.

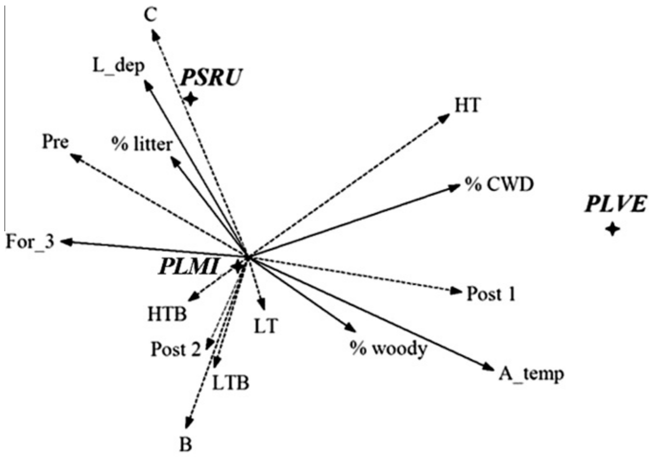

Fig. 5. Canonical correspondence analysis biplot showing relationships between the abundance of species within a lungless, terrestrial salamander assemblage and environmental, treatment, and treatment year covariates. Four-letter codes associated with the star symbol represent species responses, solid black lines represent environmental covariates, and dashed lines represent binary covariates (treatment and treatment year variables). PLMI – *Plethodon mississippi*, PLVE – *Plethodon ventralis*, PSRU – *Pseudotriton r. ruber*. Please refer to Table 1 for environmental, treatment, and treatment year covariate descriptions.

3.5. Microclimate and microhabitat comparisons

An additional objective of this study was to examine the impacts of forest management on microhabitat and microclimate characteristics in each forest stand. As mentioned in the methods, we used PCA to reduce the complexity of and to group these data into manageable habitat components. Each principal component denotes a multivariate combination of correlated variables that represents a relevant, linear gradient. Overall, we identified three ecologically-relevant components from the PCA that accounted for 78% of the overall variance in the habitat dataset. Component one (PC1) accounted for 56% of the overall variance and represented a canopy openness and groundcover habitat gradient that ranged from stands possessing lower light intensity (component score [CS] = 0.92), lower air temperature (CS = 0.87), lower soil temperature (CS = 0.86), lower slash cover (CS = 0.81), lower bare ground cover (CS = 0.76) to sites possessing lower basal area (CS = 0.92), lower canopy cover (CS = 0.87), and lower litter depth (CS = 0.63; Fig. 3). Component two (PC2) accounted for 13% of the overall variance and represented a ground cover gradient that ranged from sites possessing greater bare ground cover (CS = −0.50) to sites possessing greater herbaceous cover (CS = 0.66), greater woody cover (CS = 0.50), and greater forest level 1 cover (CS = 0.82; Fig. 3). Component three (PC3) accounted for 9% of the variance and represented a gradient of increased CWD cover (CS = 0.78) and CWD volume (CS = 0.77).

Although habitat features varied by forest stand, the random selection of forest stands resulted in similar stand features across pre-treatment environmental variables (Fig. 3). Post-treatment, we detected a significant thin and year ($F_{4,34} = 29.54$; $p < 0.0001$) and burn and year ($F_{2,34} = 7.50$; $p < 0.01$) effect for PC1 (Table 3). Specifically, control and burn-only stands had lower PC1 scores, which included lower light intensity, air temperature, and slash cover and greater litter cover compared to thin-only and thin with burn stands after treatment (Table 3 and Fig. 3). Burn-only stands tended to have lower PC2 scores, which included greater bare ground and lower herbaceous vegetation, woody vegetation, and litter cover compared to all other treatment stands after the first post-treatment year (Fig. 3). However, the difference between component two scores of burn-only stands and other treatment stands became smaller during the second year post-treatment, which was likely due to an increase in litter and vegetative cover two seasons after the original burn (Fig. 3).

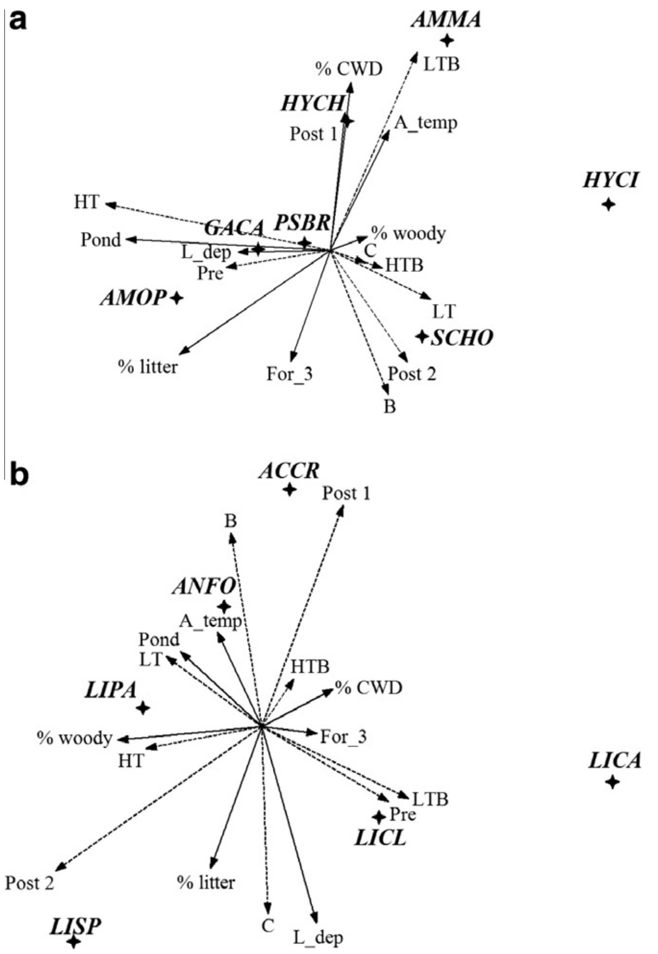

Fig. 6. Canonical correspondence analysis biplot showing relationships between the abundance of species within an ephemeral pool-breeding amphibian (a) and a semi-permanent pool-breeding amphibian (b) assemblage and environmental, treatment, and treatment year covariates. Four-letter codes associated with the star symbol represent species responses, solid black lines represent environmental covariates, and dashed lines represent binary covariates (treatment and treatment year variables). ACCR – *Acris crepitans*, AMMA – *Ambystoma maculatum*, AMOP – *Ambystoma opacum*, ANFO – *Anaxyrus fowleri*, GACA – *Gastrophryne carolinensis*, HYCH – *Hyla chrysoscelis*, HYCI – *Hyla cinerea*, LICA – *Lithobates catebeianus*, LICL – *Lithobates clamitans*, LIPA – *Lithobates palustris*, LISP – *Lithobates spenocephalus*, PSBR – *Pseudacris brachyphona*, SCHO – *Scaphiopus holbrookii*. Please refer to Table 1 for environmental, treatment, and treatment year covariate descriptions.

3.6. Species and environmental relationships

To gain a broader perspective on species responses, we used CCA to evaluate simultaneously the influence of environmental, treatment, and treatment year covariates on reptile and amphibian species assemblages (Table 2). The CCA examining the southeastern lizard assemblage explained 77% (axis 1–65%, axis 2–12%) of the species relationships displayed in Fig. 4. We detected a significant influence ($F = 2.6$; $p = 0.04$) of habitat, treatment, and treatment year covariates on lizard captures. Specifically, the horizontal axis of the biplot depicted a gradient of increasing air temperature and decreasing midstory cover (Fig. 4). Overall, we observed species-specific responses of lizards along this gradient. Specifically, *S. lateralis* captures were associated with pre-treatment, control conditions (i.e., deeper litter and closed canopy), whereas *S. undulatus* and *A. carolinensis* captures were related to increased woody vegetation cover in thin with burn stands during the second year post-treatment and higher air temperatures and CWD cover in thinned stands during the first year post-treatment, respectively (Fig. 4). These findings suggest that *S. undulatus* and *A. carolinensis* have high heliothermic requirements and colonize stands quickly after disturbance. Additionally, *P. fasciatus* and *P. lateralis* were located near the center of the horizontal axis and *S. lateralis* were located at the right side of the biplot suggesting that these species have lower heliothermic requirements compared to *S. undulatus* and *A. carolinensis* (Fig. 4). The CCA for the large-bodied upland snake assemblage explained 43% (axis 1–26%, axis 2–17%) of the species relationships displayed in Fig. 4. Although these relationships were not statistically significant ($F = 1.17$; $p = 0.34$), we detected biologically-relevant habitat, treatment, and treatment year associations. Of the seven species assessed, *C. constrictor* and Black Kingsnake (*Lampropeltis nigra*) captures were associated with higher air temperatures and increased woody vegetation cover in heavy thin stands during the second year post-treatment (Fig. 4), suggesting that canopy disturbance and relatively higher air temperatures are required to maintain these species at the stand level. *A.contortrix* captures were associated with the center of the ordination biplot, which is a common result for generalistic, frequently encountered species (Fig. 4).

In regards to amphibians, the CCA examining the terrestrial, lungless salamander assemblage explained 90% (axis 1–79%; axis 2–11%) of the species relationships displayed in Fig. 5. However, these relationships were not supported statistically ($F = 2.65$; $p = 0.14$) and may have been skewed by the relatively high number of *P. glutinosus* captured during this study, which is indicated with their location in the center of the ordination plot. Of the other two salamander species, Northern Red Salamander (*Pseudotriton r. ruber*) captures were associated with control and pre-treatment environmental conditions (e.g., greater litter depth, canopy cover; Fig. 5), suggesting that this species requires shady and cooler forested conditions. The Southern Zigzag Salamander (*Plethodon ven-*

tralis) was not detected often, which is indicated by its presence on the periphery of the CCA biplot (Fig. 5).

The CCA examining the ephemeral pool-breeding amphibian assemblage explained 55% (axis 1–32%; axis 2–23%) of the species relationships in Fig. 6. Certain species were significantly ($F = 1.90$; $p = 0.05$) associated with environmental and treatment covariates. Specifically, Marbled Salamanders (*Ambystoma opacum*) were primarily associated with forest stands located within 290 m of aquatic breeding habitats (Fig. 6). Additionally, captures of this species were highest during pre-treatment surveys in forest stands scheduled for heavy thin management that possessed pre-disturbance habitat conditions (e.g., deeper litter and greater litter coverage). Eastern Narrowmouth Toads (*Gastrophryne carolinensis*) were also weakly associated with this habitat gradient (Fig. 6), suggesting that both *A. opacum* and *G. carolinensis* may require unharvested forest conditions in close proximity to ephemeral breeding habitats. The vertical axis of Fig. 6 depicted a gradient of treatment and treatment year effects. Along this gradient, Cope's Gray Treefrogs (*Hyla chrysoscelis*) were associated with light thin with burn forest stands that possessed relatively higher CWD cover during the first year post-treatment (Fig. 6). Eastern Spadefoot (*Scaphiopus holbrooki*) captures tended to be highest in burn and light thin stands during the second post-treatment year and were not associated with a particular habitat attribute. This vertical axis of the biplot likely describes amphibian species that may benefit from habitat conditions (e.g., open canopy, road-rut pools) created by forest disturbances. Surprisingly, the CCA depicted Spotted Salamander (*Ambystoma maculatum*) captures as being highest in light thin with burn stands during the first post-treatment year. This ordination result was more likely related to the presence of a single ephemeral breeding habitat located within 290 m within one light thin with burn replicate, along with our inability to detect this species during pre- and post-treatment year two surveys in this particular stand. These effects were likely exacerbated when species captures were pooled across replicates.

The CCA examining the semi-permanent, pool-breeding amphibian assemblage explained 68% (axis 1–38%; axis 2–30%) of the variance in Fig. 6, and revealed significant ($F = 2.23$; $p = 0.02$) species relationships. In addition to a gradient displaying species captures related to a particular treatment year, species distributions were grouped along the horizontal axis in a manner that suggests the degree of water permanence of breeding sites may have influenced species captures in particular stands. American Bullfrog (*Lithobates catesbeianus*) and Green Frog (*Lithobates melanotus*) captures ordinated along the pre-treatment side of the horizontal axis, whereas Southern Leopard Frog (*Lithobates sphenocephalus*) and Pickerel Frog (*Lithobates palustris*) captures occurred together along the post-treatment side of the horizontal axis (Fig. 6). Rather than describing a treatment response, this gradient likely suggests that breeding habitats located near stands with higher captures of *L. catesbeianus* and *L. clamitans* had a longer hydroperiod, compared to the aquatic breeding habitats located near stands that had higher captures of *L. sphenocephalus* and *L. palustris*. *A. fowleri* and Northern Cricket Frog (*Acris crepitans*) captures were primarily associated with prescribed burn stands during first year post-treatment surveys. These two species were located along the middle of the horizontal axis (Fig. 6), suggesting that they require breeding sites with a different hydroperiod compared to the other species.

4. Discussion

We evaluated the collective impacts of thinning and burning forest management practices on an upland forest herpetofaunal assemblage and associated microhabitat and microclimate variables. Although our study period was relatively short (i.e., three years), our results suggest that reptiles and amphibians responded in a species-specific manner to forest restoration, potentially congruent with the physiological requirements of some species. Lizards, including *A. carolinensis* and *S. undulatus* showed immediate positive responses primarily in thin and thin with burn stands, respectively, whereas *S. lateralis* captures declined initially in all treated (i.e., non-control) stands. The CCA for the southeastern lizard assemblage supports this finding and suggests that differences in thermal properties of harvested stands versus unharvested stands were partially responsible for these patterns. Past research has found that abundance of higher-heliothermic lizards such as Six-lined Racerunners (*Aspidoscelis sexlineata*) and *S. undulatus* tend to increase following intense forest management practices including frequent (repeat) burning regimes (Mushinsky, 1985; Ruthven et al., 2008; Steen et al., in press a) and simultaneous thinning and burning management (Greenberg and Waldrop, 2008; Matthews et al., 2010). Conversely, lower-heliothermic species, such as *S. lateralis* appear to be negatively affected by intensive management through simultaneous thinning and prescribed burning (Greenberg et al., 1994). Furthermore, Vitt et al. (1998) found that higher-heliothermic lizard species tended to increase and occupy canopy gaps, whereas mid- and low-heliothermic lizard species tended to occupy gap edges and forest interior sites, respectively. Although *S. lateralis* initially responded negatively to all forest management practices in our study, we observed an increase in captures two years after treatment comparable to pre-treatment abundances, suggesting that negative impacts of forest management on this species may be short-term.

Our ability to directly examine changes in abundance for upland snake species was affected by low sample size for several species. However, we found that captures of *C. constrictor* increased significantly in heavy thin stands two years following disturbance. Although the CCA describing the response of large-bodied upland snake assemblage did not receive strong statistical support, it suggests that increased abundance of both *C. constrictor* and *L. nigra* in heavy thin plots was primarily due to relatively higher air temperatures in these stands. Management practices that increase thermal properties of the landscape including short rotation (<3 years) prescribed burns (Perry et al., 2009, 2012) and forest canopy reduction (Pike et al., 2011) have been shown to increase the abundance of certain large-bodied snake species. However, we did not observe similar increases in captures for either *C. constrictor* and *L. nigra* in thin with burn stands, which suggests that simultaneous reduction of canopy cover, litter depth, and vegetative cover may alter environmental characteristics too severely for colonization by these species. Steen et al. (2012) found that landscape scale occupancy of both *C. constrictor* and *L. nigra* was highest in open, grassy habitat patches up to 80 ha and 5 ha in size, respectively, suggesting that both of these species select habitats with an open canopy. Although *A. contortrix* was the most frequently captured large-bodied snake species in our study, we did not find support to suggest that forest management impacted this species. This is most likely due to the fact that *A. contortrix* possesses generalistic habitat use patterns at the landscape scale and inhabits a wide variety of forested habitats (Steen et al., 2012). However, Sutton (2010) showed that *A. contortrix* does select microsites that possess relatively higher amounts of CWD cover and greater litter depth compared to random microsites, which suggests that management practices that reduce these microhabitat characteristics could negatively impact *A. contortrix*. Confounding factors such as high vagility and relatively large home range size of many large-bodied snakes may have impacted our ability to evaluate the impacts of forest management on these species as we primarily used drift-fence arrays to sample herpetofauna. Future studies should consider coupling additional sampling techniques with radiotelemetry to permit comparison of abundance patterns with spatial and habitat use data to gain a broader perspective on the impacts of forest management (see Reinert et al., 2011). Our sample sizes were also too low to assess the impacts of forest management on litter-dwelling snakes, but forest harvesting has been shown to negatively influence some species in the southeastern Coastal Plain (Todd and Andrews, 2008).

Although our treatments appeared to have limited impacts on terrestrial salamanders, close examination of responses across years suggest that some species-specific responses may exist. For example, *P. mississippi* declined through all treatment years across all treatments, but the rate of change across years was not constant. Specifically, captures in thin with burn stands tended to decrease at a greater rate compared to other stands. Declines in *P. mississippi* captures throughout the study period may have been due to relatively hotter and drier environmental conditions during post-treatment sampling. Surface activity patterns of plethodontid salamanders are highly related to environmental temperature and moisture regimes (Davic and Welsh, 2004; Feder, 1983), and the reduction in rainfall amounts during post-treatment sampling in our study may have decreased detection probabilities of these salamanders across all treatments. The highest period of rainfall during our study occurred in 2005 (during pre-treatment sampling) when rainfall amounts totaled 160.0 cm (61.8 in.). In 2007, total rainfall was 92.2 cm (36.3 in.), which was 61.7 cm (24.3 in.) under the 30-year average; rainfall amounts for both 2006 and 2008 were also under the 30-year average by 18.9 cm (7.5 in.; www.ncdc.noaa.gov). However, our short sampling period of three years makes it impossible to rule out fluctuations due to natural population cycling. Hawkes and Gregory (2012) suggest that natural population cycling was the most likely hypothesis describing declines of Western Red-backed Salamanders (*Plethodon vehiculum*) over a 10-year period rather than direct impacts from timber management. The CCA also illustrated that *P. r. ruber* was associated with control stands that possessed relatively greater litter depth compared to other harvested stands, suggesting that forest management has the potential to negatively influence this species. Terrestrial salamanders in the family Plethodontidae are lungless and rely solely on cutaneous respiration, which requires the maintenance of cool and moist skin conditions (Wells, 2007). These biological adaptations may make plethodontid salamanders vulnerable to environmental disturbances that cause rapid drying of the forest litter layer (Herbeck and Larsen, 1999). Plethodontid salamanders also display species-specific temperature and moisture preferences (Feder, 1983; Grover and Ross, 2000), which makes it important to understand how management is impacting microhabitat and microclimate conditions. In a meta-analysis of plethodontid salamander responses to forest management, Tilghman et al., (2012) found that the decline of salamander populations due to forest management ranged from 29% for partial canopy removal operations to 62% for studies that examined clearcutting effects, suggesting that management practices that have a greater impact on microhabitat characterstics result in larger negative impacts on plethodontid salamanders. In our study, thin with burn treatments tended to cause the greatest changes in microhabitat and microclimate characteristics after treatment by reducing forest litter cover, litter depth, and overstory canopy cover. Based on the physiological adaptations of terrestrial lungless salamanders, reductions of these microhabitat variables are most likely to cause the greatest negative impacts to these species at our study sites over future burn cycles. However, presence of microhabitat attributes including CWD and emergent rock cover appear to mitigate negative management impacts on terrestrial plethodontid salamanders by creating moist and cool refuge sites throughout harvested landscapes (Hawkes and Gregory, 2012; Kluber et al., 2009; McKenny et al., 2006). In our study, residual CWD was piled at numerous locations throughout the treatment stands, which may have mitigated negative impacts to these species.

Relatively few studies have evaluated plethodontid salamander responses to thinning with simultaneous burning management, but it appears that negative impacts may not become apparent until burning disturbances have been repeated over several years (Matthews et al., 2010); however, Ford et al. (2010) found that repeated prescribed burning did not result in declines of salamander populations. These findings are particularly applicable for our study as our prescribed burn stands are scheduled to be burned every 3–5 years. Much uncertainty still exists regarding the impacts of repeated fire disturbances on plethodontid salamanders; therefore, monitoring should be continued at our sites over multiple burning cycles to obtain a better estimate of the impacts of these disturbances.

We found that the response of pool-breeding amphibians was related to changes in microhabitat attributes along with presence of aquatic breeding habitats located within 290 m of survey locations. Unfortunately, aquatic breeding habitats were not present at all study sites making it difficult to directly assess the impacts of forest management on these species. However, the CCA for ephemeral pool-breeding amphibians revealed that *A. opacum* and *G. carolinensis* were greatest in pre-disturbance stands located within 290 m of aquatic breeding habitats, suggesting that these species required undisturbed forest conditions and that canopy removal through forest management may negatively impact these species. Partial harvesting has been shown to have comparatively lower impacts to pool-breeding amphibians than clearcutting (Semlitsch et al., 2009), which appears to negatively influence forest-dwelling juvenile and adult pool-breeding amphibians by limiting dispersal patterns into the surrounding landscape (Fredenfields et al., 2011; Patrick et al., 2006; Popescu and Hunter, 2011). Although Veysey et al. (2009) found that ambystomatid salamanders (*A. maculatum*) were able to enter and migrate through recently harvested clearcuts during rain events, we agree with Semlitsch and Bodie (2003) in that forested buffers of at least 290 m should be established around breeding ponds to protect both aquatic and terrestrial habitats, which will ultimately permit the migration of both juvenile and adult pond-breeding amphibians throughout the landscape.

Pool-breeding amphibians that have evolved in disturbance-prone ecosystems generally benefit from the environmental conditions created by forest management (Semlitsch et al., 2009). We found that *H. chrysoscelis* captures were highest in thin with burn stands that tended to possess relatively higher CWD cover. Felix et al., (2010) found that *H. chrysoscelis* were able to colonize and breed in managed forest stands one year after harvesting. The presence of small, fishless road and skidder rut pools created during harvesting operations provide oviposition sites for some of these species (Clawson et al., 1997; Cromer et al., 2002). However, Dimauro and Hunter (2002) caution that many of these artificially created breeding habitats may function as ecological sinks because they evaporate before larvae can complete metamorphosis. In our study, the hydroperiod of aquatic breeding habitats appeared to be a primary determinant of pond-breeding amphibian captures. The contribution of these breeding habitats, whether positive or negative needs to be considered when evaluating the response of ephemeral pond breeding amphibians to forest management.

We examined herpetofaunal response to low-temperature, dormant-season (i.e., February and March) prescribed burns. We expected that prescribed burning would be more likely to negatively impact plethodontid salamanders and ephemeral pool-breeding amphibians as many species within these assemblages are surface active during these months in northern Alabama. We found that prescribed burning had minimal effects on most herpetofaunal species. This was most likely due to the fact that prescribed fires were implemented during cool, sometimes wet conditions, which resulted in low intensity and sparse burns with limited impact on litter and forest understory layers or overall forest structure. However, our results are only from one burning period, and repeated burns have been shown to impact herpetofaunal populations when initial disturbances did not trigger a response (Matthews et al., 2010). In our study, the prescribed burn stands are scheduled to be burned every 3–5 years, therefore the impacts of these burns needs to be evaluated for a longer period to understand the potential management implications of repeated prescribed burning. Past studies have indicated that the impacts of prescribed fire on herpetofauna are likely dependent on fire features (e.g., fuel loading, environmental conditions, and season of burn), and developmental life-stage of the target species (Pilliod et al., 2003). It is also important to consider whether a species or species assemblage has evolved under a fire regime when evaluating the impacts of fire or fire surrogates (Pilliod et al., 2003; Driscoll and Henderson, 2008; Russell et al., 1999; Steen et al., 2010). For example, Schurbon and Fauth (2003) illustrated that increased burning intensity led to lower overall abundance and species richness of amphibians in the Coastal Plain geographic province of South Carolina. However, their analysis included all amphibians and did not evaluate whether a species had evolved in a fire-prone ecosystem (Means et al., 2004).

Many of the forest stands examined during this study had a history of damage through *D. frontalis* infestations, which tend to be initiated by environmental stressors such as prolonged drought and overstocking of *Pinus* species (Gaines and Creed, 2003; Duncan and Linhoss, 2005). Forest damage from *D. frontalis* usually leads to an increase of standing snags, fallen logs, and large canopy gaps (Duncan and Linhoss, 2005; Knebel and Wentworth, 2007), which are beneficial to many amphibian and reptile species (Herbeck and Larsen, 1999; James and M'Closkey, 2003). Higher reptile species diversity has been observed in forest stands impacted by *D. frontalis* infestations (Sutton et al., 2010). Although we understand the importance of preventing *D. frontalis* infestations through active management strategies to avoid losing valuable timber resources, more research is necessary to determine the extent in which preexisting *D. frontalis* disturbances exacerbate amphibian and reptile reponses to prescribed burning and thinning management.

5. Conclusions

Our results suggest that one-time thinned stands may positively influence a variety of reptile species including heliothermic lizards and large-bodied upland snake species. However, as the prescribed burn stands in our study are scheduled to be burned every 3–5 years, continued monitoring of our study sites is necessary to examine the impacts of repeated disturbances on amphibian and reptile populations. Combined treatments of thinning and burning caused greater impacts to the overall microhabitat including simultaneous reduction of canopy cover and forest litter and led to increases in abundance of one reptile species (i.e., *S. undulatus*). Although plethodontid salamander populations were not differentially impacted by a specific treatment, these results should be interpreted with caution because salamander captures decreased simultaneously across all treatment plots throughout subsequent years. Due to the short study period, causes behind these declines are unknown, but relatively large departures from normal rainfall during post-treatment surveys and stochastic fluctuations due to population cycling are potential explanations that should explored further through long-term monitoring. We were unable to detect large differences between the two thinning levels on overall herpetofaunal response, suggesting that a single thin leaving 11–14 m^2/ha of residual basal area can function as a good thinning prescription to positively influence reptile species while maintaining the goals of upland hardwood forest restoration. Due to the relatively short study period (i.e., four years) and the specific forest management treatments, our study inference should be limited to xeric, upland, mixed pine-hardwood forests that have a history of disturbance through *D. frontalis* infestations in the southeastern United States. Overall, long-term studies and repeated treatments are necessary to better understand herpetofaunal community responses to prescribed burning and thinning restoration strategies.

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Appendix A

See Table A1.

Table A1
Total amphibian and reptile captures in the William B. Bankhead National Forest, Alabama, USA (2005–2008). Post-treatment year 3 data was not used in any analyses and is only included for comparative purposes. Numbers in parentheses represent recaptures. Trap nights (TN) are indicated under each sampling year. Taxonomy follows that outlined by the Society for the Study of Amphibians and Reptiles (Crother, 2012).

Species	Pre-treatment [564 TN]	Post-treatment year 1 [1086 TN]	Post-treatment year 2 [1212 TN]	Post-treatment year 3 [270 TN]	Total [3132 TN]
Frogs and toads (Anura)	101 (10)	198 (40)	254 (25)	23 (1)	576 (76)
American Bullfrog (*Lithobates catesbeianus*)	11 (0)	16 (0)	0	0	27 (0)
Barking Treefrog (*Hyla gratiosa*)	1 (0)	1 (0)	0	0	2 (0)
Cope's Gray Treefrog (*Hyla chrysoscelis*)	1 (0)	9 (0)	7 (0)	0	17 (0)
Eastern Cricket Frog (*Acris crepitans*)	0	8 (0)	3 (0)	0	11 (0)
Eastern Narrow-mouthed Toad (*Gastrophryne carolinensis*)	9 (0)	23 (1)	35 (2)	7 (0)	74 (3)
Eastern Spadefoot (*Scaphiopus holbrookii*)	4 (0)	11 (1)	35 (3)	4 (0)	54 (4)
Fowler's Toad (*Anaxyrus fowleri*)	26 (10)	88 (38)	58 (16)	5 (1)	177 (65)
Green Treefrog (*Hyla cinerea*)	0	1 (0)	4 (0)	1 (0)	5 (0)
Mountain Chorus Frog (*Pseudacris brachyphona*)	6 (0)	4 (0)	10 (0)	3 (0)	23 (0)
Northern Green Frog (*Lithobates clamitans*)	26 (0)	26 (0)	35 (3)	3 (0)	90 (3)
Pickerel Frog (*Lithobates palustris*)	7 (0)	7 (0)	32 (0)	0	46 (0)
Southern Leopard Frog (*Lithobates spenocephalus*)	9 (0)	4 (0)	52 (1)	0	65 (1)
Spring Peeper (*Pseudacris crucifer*)	0	0	3 (0)	0	3 (0)
Salamanders (Caudata)	246 (4)	325 (13)	183 (3)	30 (2)	784 (22)
Marbled Salamander (*Ambystoma opacum*)	3 (0)	1 (0)	3 (0)	0	7 (0)
Mississippi Slimy Salamander (*Plethodon mississippi*)	204 (4)	272 (13)	168 (3)	30 (2)	674 (22)
Red Salamander (*Pseudotriton r. ruber*)	15 (0)	8 (0)	6 (0)	0	29 (0)
Red-spotted Newt (*Notophthalmus v. viridescens*)	1 (0)	1 (0)	1 (0)	0	3 (0)
Southern Two-lined Salamander (*Eurycea cirrigera*)	2 (0)	0	0	0	2 (0)
Southern Zigzag Salamander (*Plethodon ventralis*)	20 (0)	37 (0)	3 (0)	0	60 (0)
Spotted Salamander (*Ambystoma maculatum*)	1 (0)	6 (0)	2 (0)	0	9 (0)
Lizards (Lacertilia)	149 (8)	259 (108)	310 (114)	62 (19)	780 (249)
Broad-headed Skink (*Plestiodon laticeps*)	16 (1)	20 (11)	29 (9)	7 (1)	72 (22)
Common Five-lined Skink (*Plestiodon fasciatus*)	22 (2)	41 (7)	35 (3)	4 (1)	102 (13)
Eastern Fence Lizard (*Sceloporus undulatus*)	12 (1)	47 (15)	70 (35)	18 (9)	147 (60)
Green Anole (*Anolis carolinensis*)	29 (4)	116 (70)	113 (64)	25 (8)	283 (146)
Little Brown Skink (*Scincella lateralis*)	70 (0)	34 (5)	61 (3)	7 (0)	172 (8)
Northern Coal Skink (*Plestiodon a. anthracinus*)	0	1 (0)	1 (0)	0	2 (0)
Southeastern Five-lined Skink (*Plestiodon inexpectatus*)	0	0	1 (0)	1 (0)	2 (0)
Snakes (Serpentes)	75 (1)	163 (2)	211 (17)	45 (4)	494 (24)
Copperhead (*Agkistrodon contortrix*)	27 (0)	68 (0)	73 (8)	10 (0)	178 (8)
Eastern Black Kingsnake (*Lampropeltis nigra*)	4 (0)	14 (0)	26 (4)	6 (0)	50 (4)
Eastern Gartersnake (*Thamnophis s. sirtalis*)	10 (0)	14 (0)	7 (0)	2 (0)	33 (0)
Eastern Hog-nosed Snake (*Heterodon platirhinos*)	0	1 (0)	4 (0)	2 (0)	7 (0)
Eastern Smooth Earthsnake (*Virginia v. valeriae*)	0	0	1 (0)	1 (0)	2 (0)
Eastern Wormsnake (*Carphophis a. amoenus*)	4 (0)	5 (0)	7 (0)	0	16 (0)
Gray Ratsnake (*Pantherophis spiloides*)	6 (0)	7 (0)	7 (0)	2 (0)	22 (0)
Midland Brownsnake (*Storeria dekayi wrightorium*)	1 (0)	0	1 (0)	2 (0)	4 (0)
North American Racer (*Coluber constrictor*)	9 (1)	35 (1)	64 (4)	16 (4)	124 (10)
Northern Scarletsnake (*Cemophora coccineum copei*)	0	1 (0)	1 (0)	0	2 (0)
Red Milksnake (*Lampropeltis triangulum syspila*)	1 (0)	3 (0)	2 (0)	0	6 (0)
Northern Rough Greensnake (*Opheodrys a. aestivus*)	0	0	3 (0)	0	3 (0)
Northern Watersnake (*Nerodia s. sipedon*)	1 (0)	1 (0)	1 (0)	0	3 (0)
Northern Red-bellied Snake (*Storeria o. occipitomaculata*)	1 (0)	2 (0)	1 (0)	0	4 (0)
Red Cornsnake (*Pantherophis guttatus*)	4 (0)	6 (1)	6 (1)	1 (0)	17 (2)
Ring-necked Snake (*Diadophis punctatus*)	4 (0)	2 (0)	3 (0)	0	9 (0)
Southeastern Crowned Snake (*Tantilla coronata*)	0	0	0	1 (0)	1 (0)
Scarlet Kingsnake (*Lampropeltis elapsoides*)	0	0	1 (0)	0	1 (0)
Timber Rattlesnake (*Crotalus horridus*)	4 (0)	4 (0)	3 (0)	2 (0)	13 (0)
Turtles (Testudines)	2 (0)	1 (0)	5 (1)	1 (0)	9 (1)
Snapping Turtle (*Chelydra serpentina*)	0	0	1 (0)	0	1 (0)
Woodland Box Turtle (*Terrapene c. carolina*)	2 (0)	1 (0)	4 (1)	1 (0)	8 (1)
All amphibians	347 (14)	523 (53)	437 (28)	53 (3)	1360 (98)
All reptiles	226 (9)	423 (110)	526 (132)	108 (23)	1283 (274)
Total	573 (23)	946 (163)	963 (160)	161 (26)	2643 (372)

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