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POTENTIAL IMPACTS OF A HIGH SEVERITY WILDFIRE ON ABUNDANCE, MOVEMENT, AND DIVERSITY OF HERPETOFAUNA IN THE LOST PINES ECOREGION OF TEXAS

DONALD J. BROWN[1], *ADAM DUARTE, IVANA MALI, MELISSA C. JONES, AND MICHAEL R.J. FORSTNER*

Department of Biology, Texas State University, 601 University Drive, San Marcos, Texas 78666, USA
[1]*Present address: Institute for Applied Ecosystem Studies, USDA Forest Service, 5985 County Highway K, Rhinelander, Wisconsin 54501, USA, e-mail: djb.ecology@gmail.com*

***Abstract.*—In September and October 2011, a high severity wildfire burned 39% of the 34,400 ha Lost Pines ecoregion in Bastrop County, Texas, USA. We assessed potential impacts of the wildfire on abundance, movement, and diversity of herpetofauna using drift fence array trap data collected prior to and after the wildfire, and anuran call survey data collected after the fire, on the 1,948 ha Griffith League Ranch. Based on *N*-mixture model analyses, abundance and movement of Six-lined Race Runners (*Cnemidophorus* [*Aspidoscelis*] *sexlineatus*) and Southern Prairie Lizards (*Sceloporus consobrinus*) were not significantly different before versus shortly after the wildfire. A capture-recapture analysis indicated that movement rates were higher in the wildfire zone for Hurter's Spadefoot Toads (*Scaphiopus hurterii*) the following spring. Based on the trap data, herpetofaunal species composition was not significantly different shortly after the fire or subsequently, during the following spring. However, the anuran call survey data indicated that anuran species richness was higher in the wildfire zone. Overall, we did not find reduced abundance or diversity of herpetofauna after the fire, a positive result for conservation in this ecoregion. In addition, our study indicated that investigations focused on fire impacts to ground-dwelling wildlife should consider detection probability when drawing inferences concerning abundances, particularly when differences in ground structure are apparent. Few other studies have examined the effects of intense fires on herpetofauna and our study indicates that herpetofaunal communities can emerge from intense fires without detrimental changes.**

Key Words.—abundance; amphibian; detection; forest; movement; reptile

INTRODUCTION

Pine-dominated forests throughout much of the United States are fire-maintained systems (Hartnett and Krofta 1989; Agee 1996; Schulte and Mladenoff 2005). In the absence of fire, flooding, and other disturbances, these forests progress towards a climax state dominated by hardwood trees (Gilliam and Platt 1999; Knebel and Wentworth 2007). Fire suppression over the last century has altered the structure and composition of historically pine-dominated forests throughout the United States (Taylor 2000; Stephens and Ruth 2005; Nowacki and Abrams 2008). One issue associated with fire suppression is a consequent increase in fuel loads, which in turn creates an environment conducive to high-severity fires (Davis 2001; Allen et al. 2002; Collins et al. 2010). Use of prescribed fire for reducing fuel loads and managing forest structure has increased dramatically over the last half century. However, much of the United States remains severely fire-suppressed, and the frequency and severity of wildfires continues to rise in many regions (Houghton et al. 2000; Shang et al. 2007; Littell et al. 2009; Miller et al. 2009).

The Lost Pines ecoregion in Bastrop County, Texas, USA, is a 34,400 ha remnant patch of pine-dominated forest thought to have become finally isolated from the East Texas Piney Woods ecoregion between 10,000 and 14,000 y ago (Bryant 1977; Al-Rabah'ah and Williams 2004). The Lost Pines was extensively logged in the 1800s and early 1900s (Moore 1977). Fire suppression was implemented throughout the ecoregion since the early to mid-1900s, causing heavy fuel loads to accumulate. On 4 September 2011, a high-severity wildfire (i.e., the Bastrop County Complex Fire) began in the Lost Pines. The fire was unstoppable due to extreme drought conditions in central Texas coupled with wind gusts in excess of 58 kph resulting from the passage of Tropical Storm Lee. After 18 d the fire was 95% contained, and the total burned area encompassed 13,406 ha. On 11 October 2011 the wildfire breached a fire break and burned another 125 ha.

Management and conservation initiatives in the Lost Pines largely focus on promoting healthy non-game wildlife populations, particularly the federally listed as endangered Houston Toad (*Bufo* [*Anaxyrus*] *houstonensis*). Although the impacts of fire on herpetofauna remains severely understudied (Russell et al. 1999; Pilliod et al. 2003), most previous research indicated that direct mortality was minimal (e.g., Cunningham et al. 2002; Fenner and Bull 2007; Radke et al. 2008; Ruthven et al. 2008). However, few previous studies assessed high-severity forest fires (Cunningham et al. 2002; Hossack and Corn 2007; Webb and Shine 2008; Chelgren et al. 2011; Hossack et al. 2012), and thus there is little information available to gauge impacts of such wildfires on abundance of herpetofauna. Because of the catastrophic nature of the Bastrop County Complex Fire, the immediate concern from agencies and landowners was the possibility of substantial direct mortality for herpetofauna and other animal taxa (Lost Pines Recovery Team 2011). In addition to direct mortality, another topic currently of interest to both managers and researchers is how herpetofaunal species detection probabilities are affected by fire. This is relevant given that fire can significantly modify ground structure, potentially influencing activity and visibility of species (Hossack and Corn 2007; Chelgren et al. 2011).

Over the past several years, we studied the use of prescribed fire for managing Houston Toad populations and other herpetofauna in the Lost Pines (Brown et al. 2011, 2014). As part of this research, we surveyed herpetofauna on our primary study property in July 2011, six weeks prior to the wildfire. Here, we use those pre-burn data to investigate potential impacts of the Bastrop County Complex Fire on abundance of several herpetofaunal species. In addition, we assess movement of herpetofaunal species shortly after the fire, as well as the following spring, and determine whether species composition differed between burned and non-burned habitat in all three sampling periods. Finally, we use anuran call survey data collected in spring 2012 to determine if species richness of calling anurans differed between burned and non-burned habitat. Collectively, these analyses provide useful insights into the potential short-term impacts of high-severity wildfires in a forest ecosystem, and the influence of detection on monitoring herpetofauna.

Materials and Methods

Study site.—We conducted this study on the 1,948 ha Griffith League Ranch (GLR), which primarily was forested with an overstorey dominated by Loblolly Pine (*Pinus taeda*), Eastern Red Cedar (*Juniperus virginiana*), and Post Oak (*Quercus stellata*), and an understory dominated by Yaupon Holly (*Ilex vomitoria*), American Beautyberry (*Callicarpa americana*), and Farkleberry (*Vaccinium arboreum*). The GLR contains three permanent ponds (i.e., ponds had not dried in at least 12 y), 10 semi-permanent ponds (i.e., ponds that typically dried several times per decade), and dozens of ephemeral pools that held water for days to months annually, depending on rainfall. Before 2009, the GLR was fire-suppressed for at least 60 y, although both prescribed burns and wildfires occurred on the study area since 2009. We conducted three low-severity prescribed burns between November 2009 and August 2010, burning ca. 378 ha. Two medium- to high-severity wildfires occurred on the GLR on 21 August 2010, which together burned 189 ha. The high-severity wildfire on 4 September 2011 burned 987 ha, followed by a second wildfire on 4 October 2011, which burned an additional 80.5 ha (Fig. 1). Because of their intensity, and thus their potential to dramatically impact the forest ecosystem, burn breaks were installed during all fires to minimize their spread.

Fire severity.—We assessed fire severity for the Bastrop County Complex Fire using vegetation plots (20 m by 50 m) randomly placed within forested habitat in 2008, and we surveyed vegetation plots between 32–66 d after the wildfire using National Park Service (2003) fire-monitoring guidelines. Of the 31 vegetation plots on

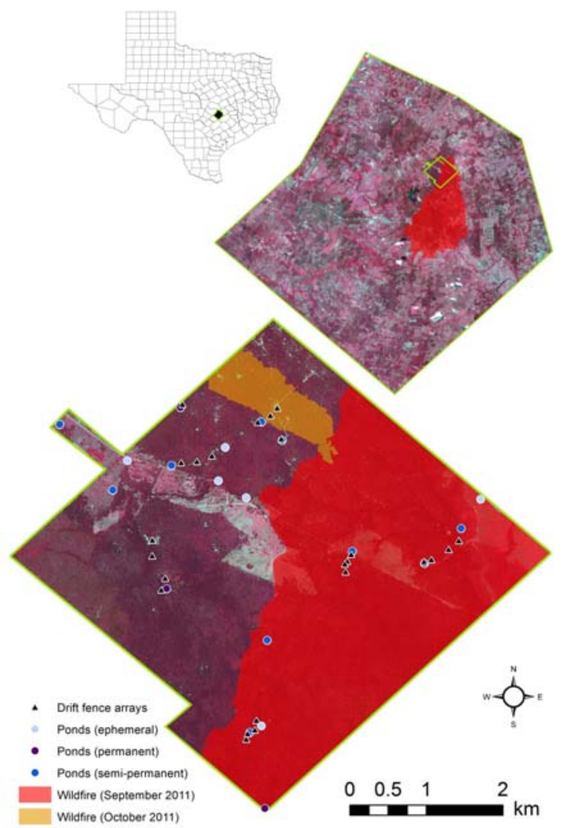

Figure 1. Aerial image of the Griffith League Ranch (GLR), Bastrop County, Texas, and its location with respect to a 13,406 ha wildfire that occurred in the Lost Pines ecoregion in September 2011, with a breach of the fire break resulting in an additional 125 ha burned in October 2011. Overlain on the image are the wildfires, drift fence arrays, and ponds used to study the abundance, movement, and diversity of herpetofauna following the wildfires. The September 2011 wildfire burned 987 ha (50.7%) of the GLR, and the October 2011 wildfire burned 80.5 ha (4.1%) of the GLR.

the GLR, 15 burned during the wildfire. We assessed burn severity to substrate within each plot using four 15 m transect lines, each consisting of four points spaced 5 m apart. We assigned each point a burn severity ranking from one (heavily burned) to five (unburned). In addition, we estimated char height and recorded status (alive or dead) for all overstorey trees (i.e., diameter at breast height > 15 cm) within vegetation plots.

***Herpetofaunal sampling*.**—We used 18 Y-shaped and six linear drift-fence arrays to trap herpetofauna. We located linear drift-fence arrays adjacent to, and parallel with, ponds to maximize amphibian captures, whereas we located Y-shaped arrays up to 600 m from the nearest pond. The aboveground height of the drift fence array flashing was at least 18 cm, with the flashing buried ca. 10 cm belowground. Y-shaped arrays consisted of three 15 m arms with a 19 l center bucket and a 19 l bucket at each arm terminus. Linear arrays consisted of a 15 m arm with a 19 l bucket at each end, and a double-throated funnel trap in the center of the array on each side of the flashing. We equipped pitfall traps with flotation devices to mitigate mortality during bucket flooding, and both pitfall and funnel traps had wet sponges to provide a moist environment. We also equipped pitfall traps with predator exclusion devices (Ferguson and Forstner 2006). Most traps within the burned area were destroyed by the fire, so we rebuilt them in their exact pre-burn locations. The burn breaks installed on the ranch during the September 2011 wildfire fortuitously resulted in a balanced sampling design for the first post-wildfire sampling period, with 12 traps (nine Y-shaped arrays and three linear arrays) located in burned and control areas, respectively. The October 2011 wildfire burned the habitat surrounding two additional arrays, resulting in 14 traps (11 Y-shaped arrays and three linear arrays) located in burned areas, and 10 traps (seven Y-shaped arrays and three linear arrays) located in control areas, for the second post-wildfire sampling period.

We trapped herpetofauna for 7 d between 16 and 23 July 2011 (hereinafter pre-burn), 7 d between 25 September and 2 October 2011 (hereinafter post-burn 1st), and 72 d between 19 February and 30 April 2012 (hereinafter post-burn 2nd). We checked traps and processed herpetofaunal captures daily, measured snout-vent length and tail length to the nearest 0.1 mm using digital calipers (Control Company, Friendswood, Texas, USA) or dial calipers (Wiha, Monticello, Minnesota, USA), and estimated weight to the nearest 0.1 g using spring scales (Pesola AG, Baar, Switzerland). We marked amphibians and lizards using toe clips (Ferner 2007), with amphibians marked individually and lizards cohort-marked.

We obtained sufficient captures of two lizard species, the Southern Prairie Lizard (*Sceloporus consobrinus*) and the Six-lined Race Runner (*Cnemidophorus* [*Aspidoscelis*] *sexlineatus*), to assess immediate wildfire impacts on abundance and detection probability. We obtained sufficient Hurter's Spadefoot Toad (*Scaphiopus hurterii*) captures and recaptures during the post-burn 2nd sampling period to determine whether detection probabilities, and thus movement rates, differed between the control and wildfire habitats using a capture-recapture analysis. Finally, we determined whether herpetofaunal species composition differed between the control and wildfire habitats during all three sampling periods.

In addition to sampling herpetofauna with traps, we also completed 24 anuran call surveys between 31 January and 16 May 2012 following the protocol of Jackson et al. (2006). On each survey night we surveyed all ponds holding water (n = 15–19). We began surveys at dusk and surveyed each pond once per survey night for 5 min. We recorded the number of individuals heard calling for all detected species, unless the number of individuals calling was too large to count accurately (typically > 10 individuals). We used these data to determine whether species richness of calling anurans (i.e., anurans engaged in breeding activity) differed between the unburned and burned locations. Thus, we assumed that a sufficient number of call surveys were completed to ensure that if a species was present at a pond, it would have been detected at least once. We believe this was likely given previous work on call survey detection probabilities on the GLR (i.e., Jackson et al. 2006), and our knowledge of anuran calling activity on the GLR based on 11 y of call survey monitoring prior to this study.

***Statistical analyses*.**—We used an *N*-mixture modeling approach to estimate herpetofaunal abundance and detection probability (*P*) prior to and shortly after the wildfire (Dail and Madsen 2011). This approach uses both spatial and temporal replication of count data to jointly estimate abundance and *P* (Royle 2004). Thus, it accounts for observed numbers being a product of both ecological and observational processes. We used the open population *N*-mixture model (i.e., pcountOpen) developed by Dail and Madsen (2011), using the software package unmarked (version 0.9-8) in program R (version 2.14.2; Fiske and Chandler 2011). We assumed our statistical populations were closed within survey periods, but open between survey periods. We used the 'constant' population dynamics model in pcountOpen (i.e., apparent survival and recruitment were not explicitly linked in the model). In addition, we included habitat (control or wildfire) and time (pre-burn or post-burn) as covariates in estimations of *P*. We used Wald tests to assess the significance of the two covariates and their interaction (α = 0.05). We determined the most appropriate distributions for our data sets (i.e., Poisson, zero-inflated Poisson, or negative binomial) by comparing the goodness-of-fit of each distribution using the 'parboot' function (Fiske and Chandler 2013. Overview of Unmarked: An R package for the Analysis of Data from Unmarked Animals. Available from http://cran.r-project.org/web/packages/unmarked/vignettes/unmarked.pdf [Accessed 5 December 2013]). We used a zero-inflated Poisson distribution for the Six-lined Race Runner analysis (P = 0.29), and a negative binomial distribution for the Southern Prairie Lizard analysis (P = 0.31).

Our data sets were not robust enough to achieve convergence when including habitat, time, and their interaction as covariates in estimations of apparent survival. Thus, to assess the short-term effect of the wildfire on abundance, we used the trap and time-specific estimated abundances as our count data in an additional analysis. We determined whether habitat, time, and their interaction were significant predictors of Six-lined Race Runner and Southern Prairie Lizard abundance using generalized linear models with Poisson distributions (Zuur et al. 2009). These analyses were conducted using the statistical package in the program R (version 2.14.2).

We conducted a capture-recapture analysis to determine whether apparent survival and P (and thus movement rates) of Hurter's Spadefoot Toads differed between burned and non-burned habitat during the post-burn 2nd sampling period. Hurter's Spadefoot Toads are explosive breeders (Wells 1977), and thus not all individuals emerged (i.e., entered the statistical population) simultaneously. Thus, we analyzed these capture-recapture data using a Cormack-Jolly-Seber model structure, which assumes the population is open (Cormack 1964; Jolly 1965; Seber 1965), and which allowed us to estimate both apparent survival and P. We specified four groups in the analysis: adult males, adult females, unburned habitat, and burned habitat. Through preliminary analyses, we determined data were too sparse to accommodate time effects on either apparent survival or P. Therefore, time was not considered as a possible constraint in subsequent analyses. Consequently, the analysis assumed apparent survival and P to be constant across the duration of the post-burn 2nd sampling period. The constraints we considered assessed whether apparent survival and P did not vary between each sex or across burned and unburned habitats (denoted *constant*), varied between each sex (*sex*), varied across burned and unburned habitats (*habitat*), or varied between each sex as well as across habitats (*sex* × *habitat*). Model selection was based on Akaike Information Criterion (AIC), corrected for small sample size (AIC_C), and AIC_C weights (i.e., the most parsimonious model had the smallest AIC_C and the largest AIC_C weight; Burnham and Anderson 1998). We conducted model averaging to estimate parameters in the event that models were competing (≤ 2 AIC_C unit deviance in the highest ranking models). We conducted this analysis using program MARK (White and Burnham 1999).

To assess whether herpetofaunal species composition differed between burned and non-burned habitat, we used an ecological distance approach (Clarke 1993). We performed Analysis of Similarity (ANOSIM) tests, using the Kulczynski ecological distance equation (Kindt and Coe 2005), for the pre-burn, post-burn 1st, and post-burn 2nd sampling periods. We included habitat as a predictor in the analyses to determine whether sites were more similar within than between burned and unburned areas. For this statistical test, the computed test statistic (R) ranges from -1 to 1, with values near 0 indicating differences in species composition between burned and unburned habitats were small. We used a permutation test (n = 1,000) to assess the significance of R (α = 0.05). For the post-burn 2nd sampling period, we did not include terrestrial juvenile anurans that metamorphosed during the sampling period in the analysis. We assumed individuals did not disperse between burned and unburned habitats within each sampling period, which was supported by our recapture data. We conducted these analyses using the software package BiodiversityR (version 1.6) in program R (version 2.14.2).

To determine whether species richness of calling anurans differed between unburned and burned areas, we used a species accumulation curve approach. This approach was appropriate because species richness estimates are affected by sample size (Kindt and Coe 2005), and the number of surveyed ponds in each habitat type was not identical (n = 9 [burned]; n = 10 [non-burned]). We calculated the average pooled species richness (± SD) at each pond, within each habitat, for each sample size (i.e., 1 to 10 ponds surveyed), and assessed differences graphically using species accumulation curves. We conducted this analysis using the software package BiodiversityR (version 1.6) in program R (version 2.14.2).

RESULTS

Fire severity.—Fire passed through 231 of the 240 substrate burn severity points during the September 2011 wildfire. The mean burn severity ranking for each plot ranged from 1.0–4.3, and the mean burn severity ranking among plots was 1.7 (Fig. 2). Of the 478 overstorey trees that were alive in plots before the wildfire, 235 were killed during the fire (49.2%). Seventy of these trees were not detected following the burn and were likely consumed by fire (i.e., 63.8% of overstorey trees were either unaccounted for or dead). Mean estimated char height for all overstorey trees was 7.3 m.

FIGURE 2. Terrestrial habitat around a drift fence array on the Griffith League Ranch (GLR), Bastrop County, Texas, USA, before (A) and after (B) a high severity wildfire on 4 September 2011. Nearly all of the understory vegetation, litter, duff, and coarse woody debris were consumed in most of the burned area, with substantial overstorey tree mortality.

Herpetofaunal impacts.—We captured 13, 9, and 18 herpetofaunal species during the pre-burn, post-burn 1st, and post-burn 2nd sampling periods, respectively. We captured 205 unique individuals during the pre-burn sampling period, 131 unique individuals during the post-burn 1st sampling period, and 7,153 unique individuals during the post-burn 2nd sampling period, of which 5,455 were recently metamorphosed Hurter's Spadefoot Toads (Table 1). The juvenile Hurter's spadefoot toads were the result of heavy rainfall-induced explosive breeding events on 10 March and 19 March 2012 (2.8 and 7.8 cm of rainfall, respectively).

Lizard abundance and detection.—We captured 45 and 97 unique Six-lined Racerunners and Southern Prairie Lizards during the pre-burn sampling period, respectively, and 30 and 31 unique Six-lined Racerunners and Southern Prairie Lizards during the post-burn 1st sampling period, respectively. We recaptured one Six-lined Racerunner and seven Southern Prairie Lizards that were marked during the pre-burn sampling period. Cumulative estimated abundance of Six-lined Racerunners was 25.5 individuals in the control habitat and 25.4 individuals in the wildfire

TABLE 1. Number of unique pre-burn (trapped 16–23 July 2011), post-burn 1st (trapped 25 September 2011 to 2 October 2011), and post-burn 2nd (trapped 19 February 2012 to 30 April 2012) herpetofaunal captures on the Griffith League Ranch (GLR), Bastrop County, Texas, USA, using 24 drift fence arrays. The post-burn 1st sample included 12 arrays located in non-burned (Control) habitat, and 12 arrays located in burned (Wildfire) habitat. The post-burn 2nd sample included 10 arrays located in non-burned habitat, and 14 arrays located in burned habitat. The relative abundance of Hurter's Spadefoot Toads (*Scaphopus hurterii*) in the wildfire habitat (post-burn 2nd) is biased low due to erosion issues following heavy rainfall events, which caused most of the wildfire habitat pitfall traps to completely fill with sand. Captured juvenile amphibians that entered the terrestrial landscape during the post-burn 2nd sampling period are shown in parentheses.

Species	Control			Wildfire[a]		
	Preburn	Post-burn 1st	Post-burn 2nd	Preburn	Post-burn 1st	Post-burn 2nd
Amphibians						
Acris crepitans (Blanchard's Cricket Frog)	1	0	0	0	0	0
Ambystoma tigrinum (Tiger Salamander)	0	0	0	0	0	59
Bufo [*Incilius*] *nebulifer* (Coastal Plain Toad)	7	0	12 (1)	3	2	25 (4)
Gastrophryne carolinensis (Eastern Narrow-mouthed Toad)	1	0	2	2	2	4
Gastrophryne olivacea (Western Narrow-mouthed Toad)	5	4	14	3	10	11
Hyla cinerea (Green Tree Frog)	0	0	0	1	0	0
Hyla versicolor (Gray Tree Frog)	0	0	0	0	0	4
Pseudacris streckeri (Strecker's Chorus Frog)	0	0	0	0	0	0 (13)
Rana [*Lithobates*] *catesbeiana* (American Bullfrog)	0	0	0	0	0	2
Rana [*Lithobates*] *sphenocephala* (Southern Leopard Frog)	1	0	2	2	0	10
Scaphiopus hurterii (Hurter's Spadefoot Toad)	0	0	468 (85)	0	0	939 (5,370)
Lizards						
Aspidoscelis sexlineatus (Six-lined Race Runner)	18	12	14	27	18	25
Sceloporus consobrinus (Southern Prairie Lizard)	77	45	42	48	33	30
Scincella lateralis (Little Brown Skink)	3	1	3	2	1	5
Snakes						
Agkistrodon contortrix (Broad-banded Copperhead)	1	0	0	0	0	0
Heterodon platirhinos (Eastern Hog-nosed Snake)	0	1	1	0	0	0
Leptotyphlops [*Rena*] *dulcis* (Texas Threadsnake)	0	0	0	0	0	1
Micrurus tener (Texas Coralsnake)	0	0	0	1	0	1
Nerodia erythrogaster (Plain-bellied Watersnake)	0	0	0	0	0	2
Storeria dekayi (Texas Brownsnake)	0	0	0	0	0	1
Tantilla gracilis (Flat-headed Snake)	1	0	1	0	1	2
Thamnophis proximus (Western Ribbonsnake)	0	0	0	1	1	0

[a]Wildfires occurred on 4 September 2011 and 4 October 2011, burning 987 ha (50.7%), and 80.5 ha (4.1%) of the study area, respectively.

habitat during the pre-burn sampling period, and 69.4 individuals in the control habitat and 68.3 individuals in the wildfire habitat during the post-burn 1st sampling period. For Six-lined Racerunners, we did not find a significant habitat-time interaction for abundance (Z = 0.03, df = 44, P = 0.977), and abundance did not differ by habitat (Z = -0.10, df = 44, P = 0.924). However, abundance differed by time (Z = -4.32, df = 44, P < 0.001). Cumulative estimated abundance of Southern Prairie Lizards was 299.6 individuals in the control habitat and 306.1 individuals in the wildfire habitat during the pre-burn sampling period, and 33.8 individuals in the control habitat and 36.6 individuals in the wildfire habitat during the post-burn 1st sampling period. For Southern Prairie Lizards, we did not find a significant habitat-time interaction for abundance (Z =

-0.23, df = 44, P = 0.819), and abundance did not differ by habitat (Z = 0.33, df = 44, P = 0.741). However, abundance differed by time (Z = 12.02, df = 44, P < 0.001).

For Six-lined Race Runners, we did not find a significant habitat-time interaction for P (Z = -0.24, P = 0.808), and P did not differ by habitat (Z = 1.63, P = 0.103), or time (Z = -1.46, P = 0.145). Estimated detection probabilities were higher post-burn in both habitats, and were higher in the wildfire habitat than the control habitat (Fig. 3a). For Southern Prairie Lizards, we did not find a significant habitat-time interaction for P (Z = -1.20, P = 0.231), and P did not differ by time (Z = 1.19, P = 0.233); however, P differed by habitat (Z = -2.76, P = 0.006). Estimated detection probabilities were higher pre-burn in both habitats, and were higher in the control habitat than the wildfire habitat (Fig. 3b).

***Hurter's Spadefoot Toad detection*.**—We captured 1,349 unique adult Hurter's Spadefoot Toads, 123 of which were recaptured at least once. The highest ranked model included no covariate for apparent survival, and habitat as the covariate for P (AIC_C weight = 0.17). However, there was little support for this model over the second highest ranked model (ΔAIC_C = 0.09), which included habitat as a covariate for apparent survival, and no covariate for P (AIC_C weight = 0.17), or additional models that included sex as a covariate (Table 2). Despite the lack of strong support for a single model, results were consistent, with both apparent survival and P for both males and females being higher in the burned than in the unburned habitat (Fig. 4).

***Herpetofaunal species composition*.**—During the pre-burn sampling period, we detected 10 herpetofaunal species in unburned habitat and 10 species in burned habitat; we captured seven species in both habitat types. During the post-burn 1st sampling period we detected four species in unburned habitat and eight species in burned habitat; we captured four species in both habitat

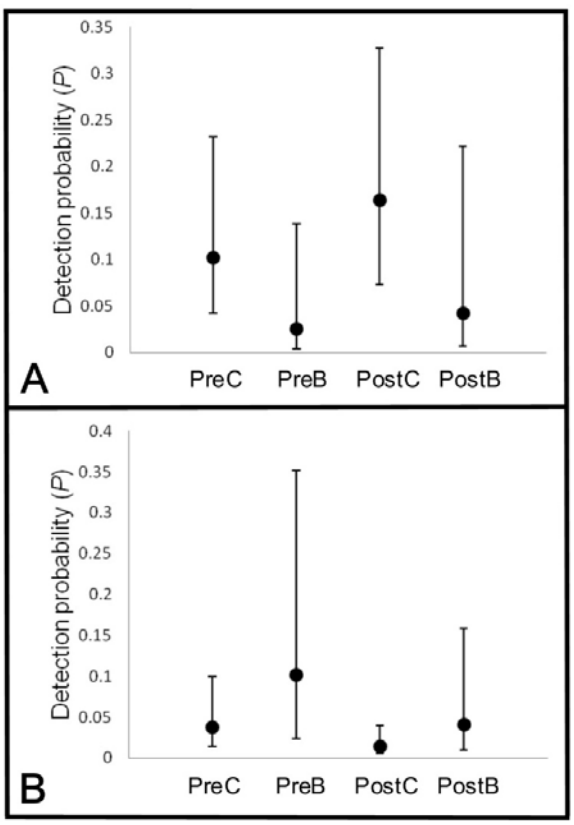

FIGURE 3. Estimated detection probabilities for (A) Six-lined Race Runners (*Aspidoscelis sexlineatus*) and (B) Southern Prairie Lizards (*Sceloporus consobrinus*), before (Pre) and after (Post) high severity wildfire in control (C) and burn (B) habitats on the Griffith League Ranch (GLR), Bastrop County, Texas, USA. Parameter estimates are shown with 95% confidence intervals. The wildfire did not appear to alter detection probabilities for these species in the short-term, which were sampled using drift fence arrays.

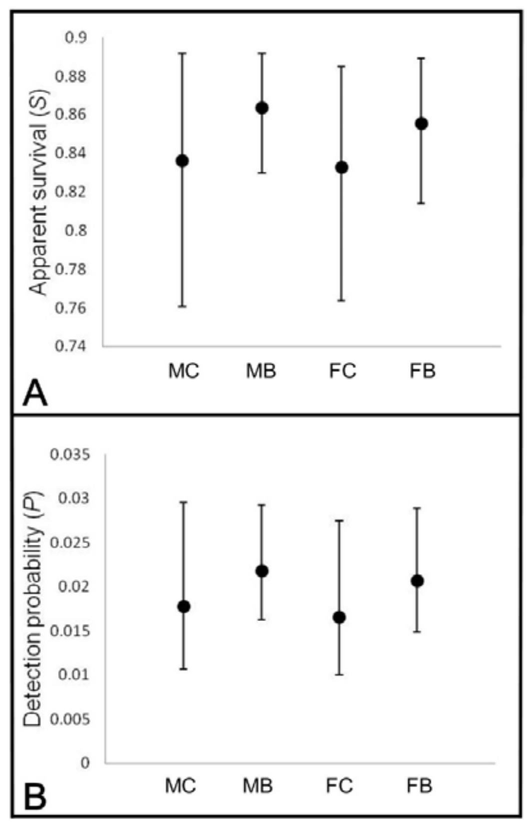

FIGURE 4. Adult male (M) and female (F) Hurter's Spadefoot Toad (*Scaphiopus hurterii*) apparent survival (S; A) and detection probability (P; B) estimates in non-burned (C) and burned (B) habitat during spring 2012 on Griffith League Ranch (GLR), Bastrop County, Texas, USA. Parameter estimates represent model-averaged results from competing models (see Table 2), and are shown with 95% confidence intervals. Estimates of both S and P were higher for both males and females in burned compared to non-burned habitat. However, support for an optimal model was low.

TABLE 2. Model selection results from a capture-recapture analysis used to determine whether apparent survival (*S*) and detection probability (*P*) varied by sex and between non-burned and burned habitat for adult Hurter's Spadefoot Toads (*Scaphiopus hurterii*) on Griffith League Ranch (GLR), Bastrop County, Texas, USA, during spring 2012. We sampled the population using 24 drift fence arrays, 14 of which were located in burned habitat. We used a Cormack-Jolly-Seber model structure, which assumes the population was open, and did not include time effects on estimates of *S* and *P*. Model selection was based on Akaike Information Criterion, corrected for small sample size (AIC_C). We used 1,349 unique individuals (742 male and 607 female) in the capture-recapture analysis.

Model	AIC_C	ΔAIC_C	AIC_C weight	Parameters
S(.) *P*(habitat)	1537.8	0.00	0.17	3
S(habitat) *P*(.)	1537.9	0.09	0.17	3
S(sex) *P*(habitat)	1538.3	0.47	0.14	4
S(habitat) *P*(sex)	1538.5	0.67	0.12	4
S(habitat) *P*(habitat)	1539.0	1.22	0.09	4
S(sex × habitat) *P*(.)	1540.0	2.23	0.06	5
S(.) *P*(sex × habitat)	1540.4	2.65	0.05	5
S(sex × habitat) *P*(habitat)	1541.1	3.31	0.03	6
S(.) *P*(.)	1541.2	3.42	0.03	2
S(sex) *P*(.)	1541.6	3.78	0.03	3
S(habitat) *P*(sex × habitat)	1541.6	3.85	0.03	6
S(.) *P*(sex)	1541.7	3.88	0.02	3
S(sex × habitat) *P*(sex)	1541.7	3.94	0.02	6
S(sex) *P*(sex × habitat)	1542.1	4.30	0.02	6
S(sex) *P*(sex)	1543.4	5.57	0.01	4
S(sex x habitat) *P*(sex × habitat)	1543.8	6.02	0.01	8

types. During the post-burn 2nd sampling period, we detected 10 species in unburned habitat and 17 species in burned habitat; we captured nine species in both habitat types. During the post-burn 1st sampling period, all species captured in unburned habitat were also captured in burned habitat. During the post-burn 2nd sampling period, the only species we captured in unburned habitat that was not captured in burned habitat was the Eastern Hog-nosed Snake (*Heterodon platirhinos*), which was represented by only one individual. We found no difference in herpetofaunal species composition between unburned and burned habitat during the pre-burn (R = -0.020, P = 0.56), post-burn 1st (R = 0.021, P = 0.32), or post-burn 2nd sampling periods (R = -0.073, P = 0.90).

***Species richness of calling anurans*.**—We found that total species richness of calling anurans was equal between ponds in unburned and burned habitat (10 total species detected). However, the species accumulation curves indicated that species richness of calling anurans was higher at ponds in burned habitat when at least eight ponds were surveyed (i.e., SD did not overlap; Fig. 5).

DISCUSSION

Most previous studies indicated mortality from fire was minimal for amphibians and reptiles (e.g., Cunningham et al. 2002; Fenner and Bull 2007; Greenberg and Waldrop 2008; Radke et al. 2008; Costa et al. 2013), and we also did not obtain evidence that mortality during a high severity wildfire was substantial. Observational studies during prescribed burns documented lizards escaping fire by burrowing under the soil and climbing trees (Bishop and Murrie 2004; Beane 2006), and Grafe et al. (2002) suggested that surface-aestivating anurans respond to auditory cues of approaching fire by seeking burn-resistant refugia. Most individuals that survived this wildfire were likely underground, given that crowning (i.e., fire in the tree canopy) was prevalent. For herpetofauna that were active at the time of the wildfire, with the possible exception of the terrestrial Three-toed Box Turtle (*Terrapene carolina triunguis*) and any dispersing semi-aquatic turtles, rapid burrowing under the soil was probably not difficult given the predominance of deep sandy soils in the Lost Pines ecoregion (Baker et al. 1979). We did find several Red-eared Slider (*Trachemys scripta elegans*) turtle shell fragments, and a partial Three-toed Box Turtle plastron within the burn zone, but cannot be certain those turtles died from the wildfire.

Our results indicated that detection probability (*P*) of two lizard species was not affected by the wildfire in the short-term. This gives some credence to results from previous studies using drift fence sampling that did not model *P* when assessing responses of lizards to fire, which as far as we are aware, includes all but one previous study (Driscoll et al. 2012). However, we note that our *P* estimates lacked precision, as indicated by our broad confidence intervals. Alternately, we obtained some evidence that *P* for Hurter's Spadefoot Toad was higher in the burned than in the unburned habitat. Chelgren et al. (2011) concluded that a wildfire positively influenced detection probabilities for five salamander species sampled using active searches. As far as we are aware, ours is the first study to suggest this phenomenon for amphibians using drift fences, which suggests that movement rates can increase after fires. However, we cannot conclude from these data what

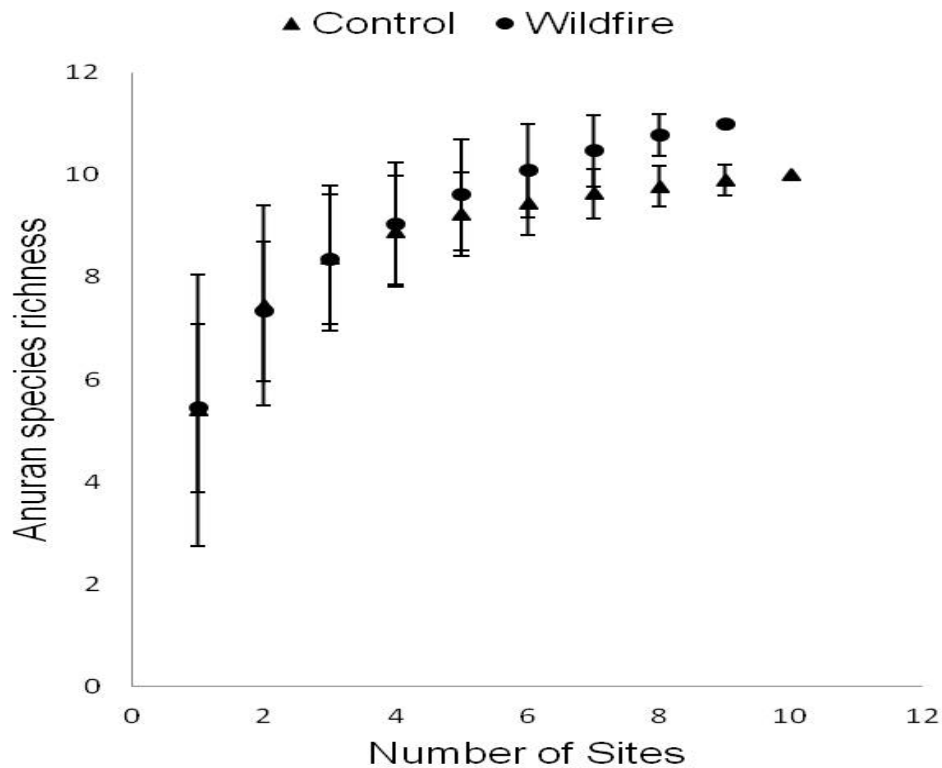

FIGURE 5. Species accumulation curves for calling anurans at ponds located within non-burned (wildfire; n = 9) and burned (control; n = 10) habitat on the Griffith League Ranch (GLR), Bastrop County, Texas, USA. We completed 24 anuran call surveys between 31 January 2012 and 16 May 2012. We surveyed all ponds holding water on each survey night (n = 15 to 19).

factors resulted in our observations, and thus we do not know if this was a general fire effect, a habitat-specific effect, or due to some other cause. We can say that weather variability was unlikely to impact our conclusions because our study design consisted of comparisons between burned and unburned habitat within-sampling periods, and thus weather effects were controlled in this study. Further, because of the timing of the fire and subsequent sampling periods, and the locations of our traps well within the burn zone, it is unlikely that our results were impacted by either differences in post-fire reproduction or movement between burned and unburned habitat. Thus, we speculate that our results indicate mortality rates were not higher in burned habitat.

An additional observation during the post-burn 2nd sampling period on adult Tiger Salamanders (*Ambystoma tigrinum*) is worth noting. Between spring 2008 and summer 2011 we trapped for 387 days, including 56 to 89 days each year between February and April, using four drift fence arrays located near a GLR pond within what would become the high severity wildfire zone (Pond 9), and captured two Tiger Salamanders. During the 72 days of trapping in spring 2012, we captured 57 unique individuals near Pond 9. These data indicate that either movement rates or movement distances increased dramatically for Tiger Salamanders following the wildfire, or alternately some other factor led to a seemingly dramatic increase in detection for this species after the wildfire.

The evidence for higher anuran species richness at ponds in the wildfire habitat indicated that anurans were not only present in burned habitat, but used it for reproduction. Thus, despite the dramatic habitat changes, the post-fire landscape was not perceived by anurans to be unsuitable habitat. This result agrees with previous studies that addressed impacts of a single wildfire on amphibian habitat use (Kirkland et al. 1996; Cummer and Painter 2007; Guscio et al. 2007; Hossack and Corn 2007). Our results indicated that pond occupancy within burned habitat may have increased after wildfire, and we intend to assess this possibility across the Lost Pines ecoregion in the coming years. Hossack and Corn (2007) found that pond occupancy by Boreal Toads (*Bufo* [*Anaxyrus*] *boreas*) increased dramatically following a wildfire in Montana. If the endangered Houston Toad responds similarly in the Lost Pines, this could be viewed as a positive effect of wildfire. During spring 2012 anuran call surveys throughout the Lost Pines ecoregion, we detected Houston Toads at ponds in both unburned and burned habitat, including several ponds within the burn zone where we had not detected them during call surveys over the last decade. However, we note that because this species has critically low numbers (Duarte et al. 2011), and Houston Toad breeding success at a given pond increases exponentially with number of calling males (Gaston et al. 2010), increased occupancy in suboptimal habitat (e.g., ponds located within residential areas or similar small forest fragments) may actually expedite the extirpation of this species from the Lost Pines.

If there is a lack of negative wildfire impacts on herpetofaunal abundance and diversity, as suggested from our analyses, then it is a positive result for management of the endangered Houston Toad and other threatened herpetofauna (i.e., Texas Horned Lizard, *Phrynosoma cornutum*, and Timber Rattlesnake, *Crotalus horridus*) found in the Lost Pines (Brown et al., in press). Despite the lack of evidence for negative short-term wildfire effects on herpetofauna found in this study, indirect effects (e.g., prey reduction, litter and coarse woody debris removal, overstorey tree thinning, water quality impacts) are still a major concern (Brown et al. 2014). In our opinion, the most critical impact of the wildfire was overstorey tree loss. Both the endangered Houston Toad and threatened Timber Rattlesnake prefer heavily canopied environments (U.S. Fish and Wildlife Service 1984; Brown 1993). The estimated loss of 63.8% of overstorey trees on the GLR was similar to an overall projection from the Texas Forest Service for the entire burned area (78%; Lost Pines Recovery Team 2011), and it will take decades for these areas to return to mature forest.

Although most of the remaining wild Houston Toads are currently in the Lost Pines ecoregion (Brown 1971, 1975), it would be valuable to focus some future recovery efforts (e.g., headstarting; Vandewege et al. 2012) on other geographically disjunct populations to prevent against extinction in the wild, given the unknown trajectory of the Lost Pines population. This latter point is supported by previous studies emphasizing that probability of extinction in the Lost Pines will likely increase dramatically with a catastrophic wildfire (Seal 1994), or lack of multiple viable populations (Hatfield et al. 2004).

In conclusion, we found no evidence that a high intensity wildfire was detrimental to amphibians and reptiles, even in the immediate short term, a result consistent with most of the existing literature on the effects of intense fires on herpetofauna (e.g., Cunningham et al. 2002; Hossack and Corn 2007). Because wildfires may result in long term increases in herpetofaunal diversity due to habitat diversification (McCoy et al. 2013; Smith et al. 2013), our study strengthens the argument that fire can be an appropriate management tool for herpetofaunal communities. Further work monitoring herpetofauna in the Lost Pines to document population and community trends in relation to natural forest recovery and active restoration initiatives is necessary to allow us to more thoroughly understand the dynamics of this system.

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DONALD J. BROWN received a B.Sc. in Fisheries & Wildlife-Wildlife Specialization at the University of Minnesota-Twin Cities, a M.Sc. in Wildlife Ecology at Texas State University, and a Ph.D. in Aquatic Resources at Texas State University. He is currently a Postdoctoral Research Associate at the University of Wisconsin-Madison. Donald is primarily interested in research that contributes to management of wildlife populations. His current research is focused on climate change impacts and management of the endangered Kirtland's Warbler (*Setophaga kirtlandii*), and Wood Turtle (*Glyptemys insculpta*) ecology in Minnesota. (Photographed by Jay Janner)

ADAM DUARTE received a B.Sc. in Wildlife and Fisheries Sciences at Texas A&M University and a M.Sc. in Wildlife Ecology at Texas State University, and is currently a Ph.D. candidate in the Department of Biology at Texas State University. Adam is primarily interested in applied research that assists with conservation and management efforts for wildlife populations. Although he is currently involved in research on a range of taxa, his dissertation research investigates population ecology for the federally listed endangered Golden-cheeked Warbler (*Setophaga chrysoparia*). (Photographed by Mark A. Ricca)

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IVANA MALI is a Ph.D. candidate in the Department of Biology at Texas State University-San Marcos. She is primarily interested in sustainable harvest of freshwater turtles. She is currently involved in assessing freshwater turtle populations in Texas and evaluating current harvest regimes. (Photographed by Bei DeVolld)

MELISSA C. JONES received a B.Sc. in Zoology and a M.Sc. in Wildlife Ecology from Texas State University. She is currently a Ph.D. candidate in the Aquatic Resources program at Texas State University. Melissa is interested in conservation and management of endangered herpetofauna. Her research primarily focuses on habitat suitability, density dependence parameters and head-starting of the endangered Houston toad (*Bufo* [*Anaxyrus*] *houstonensis*). (Photographed by Kimberly Burge)

MIKE FORSTNER is a Professor in Biology at Texas State University, and the Alexander-Stone Chair in Genetics. He has a B.Sc. from Southwest Texas State University, M.Sc. from Sul Ross State University, and a Ph.D. from Texas A&M University. He has broad interests in the effective conservation of rare taxa, particularly reptiles and amphibians. The students and colleagues working with him seek to provide genetic and ecological data relevant to those conservation efforts. (Photographed by James Stout)