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Response of Two Terrestrial Salamander Species to Spring Burning in the Sierra Nevada, California

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Abstract—Terrestrial salamanders may be vulnerable to prescribed fire applications due to their moist, permeable skin and limited mobility. We present data collected on terrestrial salamander populations in a ponderosa pine-dominated forest in the Sierra Nevada where fire was applied in the spring. Two species, Sierra ensatina (*Ensatina eschscholtzi platensis*) and gregarious slender salamander (*Batrachoseps gregarius*), were captured under coverboards. Capture rates of ensatinas declined within the first 2 years after burning, but postfire captures were similar to or greater than capture rates on unburned plots. Capture rates of slender salamanders were more variable, but high capture rates on burned plots suggest they persist following fire. We captured fewer small ensatinas within 2 years of burning, but sizes of slender salamanders pre and post burning were similar. Salamanders were captured in both closed and open canopy forests, and presence under individual coverboards was associated with deeper litter and greater canopy closure. Coverboards may be avoided for a year or more by gregarious slender salamanders, and capture rates were highest during winter and early spring. Though sample sizes were small and conclusions should be made with caution, results indicated no strong adverse effects from spring burning. Suitable habitat may have been maintained by the patchy burn pattern characteristic of spring burns.

Introduction

Salamanders play important roles in forested ecosystems. They are long-lived, have low mobility, and are sensitive to environmental changes, thus they are good indicators of ecosystem health (Davic and Welsh 2004; Russell and others 1999; Welsh and Droege 2001). Terrestrial salamanders (Plethodontidae), in particular, require moist conditions and are vulnerable to land management activities, such as prescribed burns and logging, that alter forest floor environments (Welsh and Droege 2001). The study of terrestrial salamanders is important to understanding the impact of these activities and making informed management decisions.

Fire can alter populations both through direct mortality and through indirect effects via habitat alteration. Information on salamander response to fire is limited and primarily restricted to the southeastern United States (Pilliod and others 2003). Although little information is available, direct mortality is thought to be fairly low (Renken 2006; Russell and others 1999). Individuals may find refuge underground because the majority of many salamander populations are subterranean (Bailey and others 2004; Petranka and Murray 2001; Taub 1961). On the other hand, high susceptibility to fire has been suggested because terrestrial salamanders have small home ranges, move slowly, and are probably limited in their ability to disperse (Kleeberger and Werner 1982; Staub and others 1995). Studies of prescribed fire's effects in the southeastern United States found no changes in relative abundance of terrestrial salamanders (Ford and others 1999; Keyser and others 2004; Kilpatrick and others 2004; Moseley and others 2003), whereas salamander numbers were lower in forests where prescribed fire and thinning were applied in Maryland (McLeod and Gates 1998). Studies of prescribed fire in western North America are more limited and have generally been conducted in coastal climates that host different salamander communities and natural fire regimes compared to interior forests. No changes in relative abundance of slender salamanders (*Batrachoseps* sp.) were found for prescribed fire in coastal California, (Vreeland and Tietje 2002). However, in coastal Oregon, some salamander species declined from sites that were burned and clearcut (Cole and others 1997).

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Insights into salamander response to prescribed fire may be available from wildfire studies, but differences in fire season and severity are likely to alter response. Following a New Mexico wildfire, no effect on salamander presence was observed, but there was a shift to smaller size classes in a terrestrial salamander (Cummer and Painter 2007).

Prescribed fire is commonly used in coniferous forests of the western United States where decades of fire suppression, climatic changes, and other types of disturbance have altered forest structure and natural fire regimes (Pyne 1984). Fire regime changes in California's Sierra Nevada have generally increased tree and shrub densities, as well as lengthened fire return intervals and increased fire severities (McKelvey and others 1996). Historically, fires on the west slope of the Sierra Nevada occurred late in the growing season from mid-summer to early fall (Caprio and Swetnam 1995), but high fuel levels and air quality limitations have led managers to prescribe fires during periods of high moisture, including in the spring. On one hand, these periods coincide with terrestrial salamander emergence from subterranean habitats, creating a higher risk of direct mortality (Pilliod and others 2003). On the other hand, because these fires are generally less severe and burn incompletely, habitat alteration is small compared to contemporary wildfires.

Many salamander species in the Sierra Nevada are endemic and little is known about their population status (Jennings 1996) much less fire's affect on the species' populations or habitats. Because many studies lack replication and pre-fire data (Russell and others 1999), we collected data on salamanders before and after prescribed burns on multiple plots. We present data collected on terrestrial salamander populations up to 6 years following burning in the Sierra Nevada, California, where prescribed fire was applied in the spring. We also measured individual size because differences in prey size or probability of surface activity can lead to size-dependent fire response (Cummer and Painter 2007). For instance, small salamanders have a larger surface area to volume ratio and may be more vulnerable to disturbance (Hairston 1987). Additionally, a greater proportion of small (young) individuals may indicate that individuals are using suboptimal habitats (Welsh and others 2008). We also collected data on physical characteristics of coverboard arrays to investigate their influence on capture rates.

Methods

The study took place from 2001 to 2004 in the Sierra National Forest, Fresno County, California, at elevations ranging from 1,000 to 1,400 m, approximately 65 km east of the city of Fresno (37°02'N, 119°15'W). The forests of the study area were dominated by ponderosa pine (*Pinus ponderosa*) but also contained canyon live oak (*Quercus chrysolepis*), black oak (*Quercus kelloggii*), incense cedar (*Calocedrus decurrens*), sugar pine (*Pinus lambertiana*), and white fir (*Abies concolor*). Ground cover consisted primarily of a thick layer of pine needles interspersed by mountain misery (*Chamaebatia foliolosa*), a common perennial ground-cover species. Forested areas were intermixed with granitic outcrops and shrub fields dominated by whiteleaf manzanita (*Arctostaphylos viscida*). Precipitation fell primarily during the winter in the form of rain or snow. Cumulative precipitation from January through May at the Pine Flat Dam (National Weather Service Cooperative Station 46896, approximately 24 km from the study area) was 30.7 cm, 17.8 cm, 24.6 cm, and 19.1 cm from 2001 to 2004, respectively. The average precipitation for the preceding 10 years was 41.9 cm.

The study area consisted of six plots, four of which the U.S. Forest Service burned during the study for its fuel management programs. Prior to 1997, the most recent fire in the area was in 1941. Two plots were burned in April of 2002. Two plots that were burned in April 1997 and May 1998, respectively, were burned again in June 2003. Thus, there were two types of plots that were burned once: those burned in 2002 and those burned in 1997/1998. In 2003, the plots burned in 1997/1998 were burned again (table 1). The remaining two plots remained unburned. Three plots (one in each treatment type) were on the Rush Creek drainage and three were approximately 4 km away on the Big Creek drainage, both of which flow into Pine Flat Reservoir on the Kings River.

Surveys for terrestrial salamanders were conducted from 2001 to 2004 using coverboard arrays (DeGraaf and Yamasaki 1992). Boards were made of 30- by 30- by 2.5-cm plywood and left to weather outside at least two months before placement (Grant and others 1992; Monti and others 2000) at study locations in early 2001. Boards were arranged on each of the 6 plots in 3 separate arrays of 18 boards each (324 boards total). The boards were placed 12 m apart based on

Table 1—Numbers of salamanders captured under coverboards for survey days from 2001 to 2004. Numbers are grouped by burn history with 2 replicate plots per treatment (control, burned once, burned twice) and 3 arrays of 18 boards each. Diagonal hatching indicates surveys done after plots were burned once and cross-hatching after plots were burned twice. Where no hatching appears, no fire had been recorded for greater than 50 years.

Year		**2001**	**2002**	**2003**	**2004**
Number of survey days		**4**	**2**	**5**	**2**
Sierra ensatina	**Plot**				
(*Ensatina eschscholtzi platensis*)					
Control	U1	1	0	6	2
	U2	1	0	2	1
Burned 2002	B1	5	0	5	1
	B2	3	0	5	1
Burned 1997/98 and 2003	P1	0	1	5	0
	P2	0	1	9	2
Gregarious slender salamander					
(*Batrachoseps gregarius*)					
Control	U1	0	0	0	0
	U2	0	0	1	0
Burned April 2002	B1	0	1	14	1
	B2	0	0	0	0
Burned April 1997 and June 2003	P1	0	1	8	1
Burned May 1998 and June 2003	P2	0	0	0	0

home range estimates (Staub and others 1995) to maximize unique individuals available for capture. Arrays were generally rectangular and near creeks or seeps. We placed two arrays on each plot in a closed canopy forest and one in an open forest dominated by whiteleaf manzanita because we did not know specific habitat associations. We made a shallow depression underneath each board to create a space for the salamanders. Litter was removed during initial placement so boards were in contact with the ground, but thereafter litter was allowed to accumulate on the boards (fig. 1). We recorded the general characteristics of each array before fire, including the slope and aspect of the overall terrain and distance to permanent water, which was measured from the center of each array using ArcMAP v.8.1 (Environmental Systems Research Institute, Inc., Redlands, California). Litter depth (mm) and canopy closure (%) were measured at each board. Canopy closure was calculated by averaging cover values obtained with a spherical densitometer from the four cardinal directions while standing at the edge of the coverboard and facing away from the center.

We conducted surveys of all boards on 6 plots on each of 13 days, November through July from 2001 to 2004. We conducted surveys at least two weeks apart, usually the day after rainfall when the surface soil was moist and salamanders were expected to emerge (Fellers and Drost 1994; Marsh and Goicochea 2003). Surveys were not conducted during rainfall (Jaeger 1980). Winter snow and dry conditions from June to October limited our survey opportunities. We removed boards prior to burning and replaced them immediately following fire on one plot in 2002, but found low fire intensities made it unnecessary. We replaced lost or damaged boards with boards of the same age that had been stored outside to weather equally.

We identified the species of each captured salamander and measured from the tip of the snout to the posterior tip of the vent (i.e., snout-vent length or SVL) (Petranka 1998). We marked individuals of Sierra ensatina with a visible implant elastomer (Northwest Marine Technologies) for individual identification (Donnelly and others 1994). Marks were placed on the belly close to the limbs in three places with two of four possible colors. Marking slender salamanders in the field proved to be difficult and was abandoned.

We calculated capture rates of salamanders as the number of salamanders per survey effort (captures per 100 coverboard survey days). Capture rates were used to evaluate survey timing, differences in array placement by habitat, and burn effects. We note that captures and comparisons are for the "visible population" and acknowledge that the majority of the population is subterranean (Taub 1961). Results were

Figure 1—Coverboard lifted with adult Sierra ensatina (*Ensatina eschscholtzi platensis*) underneath (photo by Karen E. Bagne). Photo was scanned from 35 mm film and was adjusted for brightness and contrast.

not analyzed statistically because capture rates per treatment were low. Appropriate statistical analysis for the study design would require a time by treatment interaction, which would further reduce the available degrees of freedom. We present all data and discuss capture rates and mean salamander size related to burning qualitatively.

We modeled salamander presence under individual boards using a generalized linear model with a binomial distribution to investigate factors related to habitat and topography (PROC GENMOD in v.8.01, SAS Institute, Inc., Cary, North Carolina). We tested litter depth and average canopy closure in this model with a repeated measures statement identifying the array as the subject. Measures related to arrays (i.e., slope and distance to water) were also compared using generalized linear models with the repeated subject designated as the plot. Variables were considered to be related to salamander presence when $P < 0.05$ for the parameter estimate from the generalized estimating equation (GEE), which fits models to the correlated responses.

Results

Fires were generally of low intensity and left a patchwork of burned and unburned areas. Few of the boards left in place during burning were consumed. Of these nine burned arrays, three lost one board, one lost two, and one lost three.

We found 78 terrestrial salamanders of two species: 51 Sierra ensatina and 27 slender salamanders (table 1). Recapture rates of Sierra ensatina were low with nine recaptures of six individuals over four

Figure 2—Gregarious slender salamander (*Batrachoseps gregarious*) after capture under coverboard (photo by Karen E. Bagne). Photo was scanned from 35 mm film and was adjusted for brightness and contrast.

years. Slender salamanders were identified as gregarious slender salamanders (*Batrachoseps gregarius*) based on the location of the study area and physical characteristics of the species as described by Jockusch and others (1998) (fig. 2). A single adult California newt (*Taricha torosa sierrae*) was the only other salamander captured. The only other vertebrates we found were three Western fence lizards (*Sceloporus occidentalis taylori*) and one Northern alligator lizard (*Elgaria coerulea palmeri*).

Though we conducted surveys on various dates when soils were moist, captures in March (3.7 captures per 100 coverboard survey days) and April (2.5 captures per 100 coverboard survey days) were the highest (fig. 3). Capture rates were lower in May (1.2 captures per 100 coverboard survey days) and only two salamanders were captured on each of our survey days in July, November, and December. In addition, salamander presence under boards increased over time, but no gregarious slender salamanders were captured in 2001 (table 1).

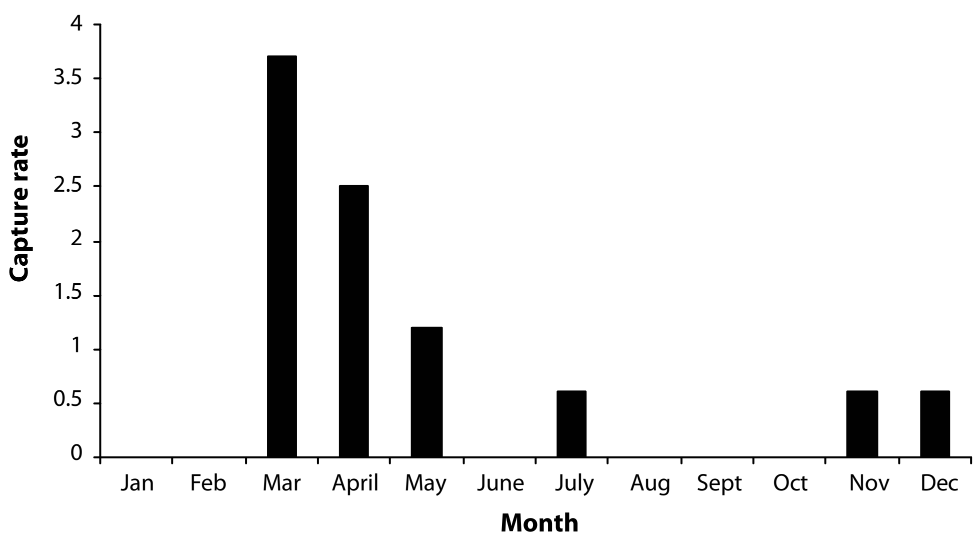

Figure 3—Capture rates for all salamanders combined by month for 2001 to 2004. Capture rate is number of captures per 100 coverboard survey days. Months where no capture rate appears had no surveys rather than no salamanders.

Table 2—Mean (SE) physical characteristics of boards where terrestrial salamanders were present (n = 268) or absent (n = 56) over the entire study period 2001 to 2004. *P*-values are from generalized linear models testing differences in salamander presence.

Variable	Present	Absent	*P*
Slope (%)	17.9 (1.4)	15.0 (0.5)	0.58
Distance to water (m)	62.4 (8.5)	79.2 (4.0)	0.35
Litter depth (mm)	53.8 (4.5)	41.0 (2.0)	0.03
Average canopy closure (%)	89.6 (1.2)	81.6 (1.2)	0.002

Table 3—Number and capture rate for salamanders found under coverboards for survey days 2001 to 2004. Capture rate is number of captures per 100 coverboard survey days. P1, burned April 1997, and P2, burned May 1998, were burned again in 2003, but post-2003 burn data are excluded here.

		Sierra ensatina (*Ensatina eschscholtzi platensis*)		Gregarious slender salamander (*Batrachoseps gregarius*)	
Burn status	**Plot**	**Number**	**Capture rate**	**Number**	**Capture rate**
Unburned	U1	9	1.28	0	0
	U2	4	0.57	1	0.14
1-2 years postburn	B1 pre	5	2.31	0	0
	B1 post	6	1.23	16	3.29
	B2 pre	3	1.39	0	0
	B2 post	6	1.23	0	0
3-6 years postburn	P1	6	1.01	9	1.52
	P2	10	1.68	0	0

The percentage of canopy closure differed between the two sampled habitats, averaging 90.9% (SD = 7.1) for closed forest coverboards and 67.0% (SD = 24.9) for open forest coverboards. Gregarious slender salamander capture rates between the two habitats were similar, with 0.6 vs. 0.7 captures per 100 coverboard survey days for closed and open forest, respectively. More Sierra ensatinas were captured in closed forest arrays than open ones (1.5 vs. 0.6 captures per 100 coverboard survey days). Salamander presence at arrays was not related to slope (deviance = 1.06, Z = -0.55, P = 0.58; table 2) or distance to water (deviance = 1.06, Z = 0.93, P = 0.35; table 2) though the maximum distance for any array was 225 m. Terrestrial salamanders were found under boards with greater litter depth (deviance = 0.99, Z = 2.16, P = 0.03; table 2) and increased canopy closure (deviance = 0.96, Z = 3.12, P = 0.002; table 2). Litter depth and canopy closure were positively correlated (r^2 = 0.17).

Captures varied considerably by plot. Gregarious slender salamanders were only present on plots in the Big Creek drainage and none were found in the Rush Creek drainage (table 1). Captures of Sierra ensatinas declined after burns in 2002 (table 3), but preburn capture rates for these two plots were higher than those of the unburned plots, and postburn capture rates were similar to the unburned plots. Postburn capture rates after 2002 burns were similar to those on plots 3 to 6 years postburn (table 3). No gregarious slender salamanders were captured on burned plots prior to burning, but postburn capture rates in 2002 were greater than captures on the plot burned in 1997 or the unburned plot (table 3). We captured too few salamanders on the two surveys following the second burn in 2003 to evaluate the effects of burning twice (table 1). However, both Sierra ensatinas and gregarious slender salamanders were present after the second burn.

The mean size of captured salamanders also varied by plot. Results from plots burned in 2002 suggest the mean size of Sierra ensatinas increased after burning, indicating fewer small salamanders (table 4). Mean size of gregarious slender salamanders was similar on plots of all burn histories (table 4).

Discussion

Although our study design provided for pre and postburn data collection, low counts were problematic

Table 4—Mean salamander snout-vent length (mm) (SVL) and standard errors (SE) by burn status and plot. Three- to six-year data from plots that were burned again in 2003 are excluded.

		Sierra ensatina (*Ensatina eschscholtzi platensis*)		Gregarious slender salamander (*Batrachoseps gregarius*)	
Burn status	**Plot**	**Number**	**Mean SVL ± SE**	**Number**	**Mean SVL ± SE**
Unburned	U1	4	46.3 ± 6.6	0	0
	U2	9	53.2 ± 6.7	1	32.9
1-2 years postburn	B1 pre	5	41.0 ± 8.4	0	0
	B1 post	6	59.0 ± 5.5	15	31.2 ± 1.0
	B2 pre	3	34.4 ± 3.8	0	0
	B2 post	6	37.8 ± 5.1	0	0
3-6 years postburn	P1	6	34.4 ± 5.9	8	32.3 ± 1.3
	P2	10	51.5 ± 5.3	0	0

in that they prevented both evaluation of population changes over time and using recapture data to estimate detectability or population size. While we recognize the limitations of the data, we know of no other published studies on the effects of fire on salamanders from the Sierra Nevada. Thus, these data are a first step toward filling a gap in our understanding of prescribed fire effects on terrestrial salamanders and offer insights to aid future study design. To our knowledge, no information outside of taxonomic studies has been published on gregarious slender salamanders.

Both Sierra ensatinas and gregarious slender salamanders persisted after burn applications up to 6 years following fire. Despite initial concerns about vulnerability, salamander numbers were not lower on burned sites. Capture rates of Sierra ensatinas declined in the first two years after burning but were similar to unburned plots and plots 3 to 6 years postburn. Only one gregarious slender salamander was caught on unburned plots and none were captured in the preburn period, but high capture rates on burned plots indicate that these salamanders persisted following fire.

We are cautious in our conclusions, but feel there is enough evidence to conclude that low intensity spring burning did not significantly harm terrestrial salamanders up to 6 years postfire. These results are consistent with others who have found neutral responses to fire despite differences in the studied fire intensities (Ford and others 1999; Keyser and others 2004; Kilpatrick and others 2004; Moseley and others 2003; Vreeland and Tietje 2002).

It is possible that differences in capture probabilities related to fire were responsible for the patterns in capture rates we observed. Coverboards may be more attractive and have greater probability of capture following fire if natural cover is reduced, but capture rates were lower or unchanged on plots immediately following fire. Additionally, differences in coverboard capture rates following fire may not represent a change in numbers, but a change in surface activities. Temperature and surface moisture are known to affect surface activity and consequently capture rates (Petranka and Murray 2001). Our captures increased in years of higher precipitation, but we were able to reduce this bias by completing surveys on all plots within a single day when conditions were similar for burned and unburned plots. Differences in surface activity related to fire may explain why, on the plot where capture rates were most reduced following burning, we also found an increase in mean size. Smaller salamanders may be captured less often at the surface following fire if they are more sensitive to surface conditions and if fire reduces favorable conditions. Fire's affect on surface activity needs further investigation.

Based on our findings, we can make a number of recommendations for future studies. Though we did not investigate alternative sampling methods, we felt the use of coverboards was generally successful. Surveys could be completed in one day by two people, the survey area was consistent in space and time, and habitat disturbance was minimal. Numbers of individuals captured were low overall (1.85 captures per 100 coverboard survey days), but capture rates were similar to those reported by Vreeland and Tietje (2002) from the California oak woodlands (1.74 captures per 100 coverboard survey days). Although spring and fall are common survey months, at this location capture rates were highest during late winter and early spring with few captures in the fall (fig. 3). Low precipitation during most of the study period may have contributed to low capture rates.

With these capture rates and the very low ensatina recapture rate, we concluded the number of sampling days and/or coverboards needs to be increased. We were limited by funding and winter access to our study area, which may be improved in other studies, but we were also limited by the restricted period of precipitation in a Mediterranean climate. We did not sample in January or February, but these may also be good sampling periods for this climate considering our high capture rates in March. Although replication of plots and arrays helped reduce bias related to plot differences, the strong differences between capture rates highlights the need for replication. Gregarious slender salamanders avoided coverboards for at least one year after installation, longer than for Sierra ensatinas or reported for other species (Monti and others 2000). This should be taken into account when initially placing boards. Based on captures by habitat, captures in this region can be maximized by sampling only closed canopy forest, but because both salamander species were also captured in open forest mixed with whiteleaf manzanita, the suitability of this habitat should be noted for management.

Association of salamander presence with litter depth and persistence after spring fire indicates that spatial heterogeneity in burning may be an important factor in predicting effects. Fire intensities were low, resulting in incomplete consumption of the litter layer and logs as well as limited affect on canopy closure. As a result, suitable terrestrial salamander habitat was retained. Although high intensity fires would likely have a different impact, low intensity burns are typical of management burning on National Forest land in the Sierra Nevada where high fuel loads and air quality issues limit burning to moist times of year. Our findings are encouraging evidence for the persistence of terrestrial salamanders during spring management burning up to 6 years postfire. Finally, terrestrial salamanders are still threatened by climate change (Parra-Olea and others 2005), disease (Byrne and others 2008), and other forest management activities such as logging (Herbeck and Larsen 1999; Petranka and others 1993) and road building (deMaynadier and Hunter 2000), and deserve more active study.

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