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EFFECTS OF PRESCRIBED FIRE ON *COLUBER CONSTRICTOR MORMON* IN COASTAL SAN MATEO COUNTY, CALIFORNIA

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Abstract.—**Although prescribed burns are used in many locations where reptiles are present, the effects of fire on reptiles are poorly understood. Our objective was to determine the effects of prescribed fire on the Western Yellow-bellied Racer, *Coluber constrictor mormon*, at a study site in coastal San Mateo County, California. We sampled an unburned site and an adjacent prescribed burn site two to five years post fire in the spring and summer of 2007 through 2010. We found no differences in apparent survival or body condition of snakes between burned and unburned sites. The burned site had lower snake abundance than the unburned site in 2008 but we found no difference in abundance between sites in 2009 and 2010. We also found a difference in body condition across years and asymmetry in the probability of transition (movement between burned and unburned sites). Vegetation cover between the burned and unburned sites was similar. This study serves as a baseline for investigation of the effects of fire on *Coluber constrictor mormon* at our study site and provides further insight on the effects of prescribed fire on reptile species.**

Key Words.—burn; coastal prairie; mark-recapture; multi-state model; reptiles; snake; survival; Western Yellow-bellied Racer

INTRODUCTION

Fire is essential to the maintenance of plant community composition and structure in many ecosystems (Barbour et al. 2007; Carle 2008). The use of prescribed fire to control vegetation in coastal California can be traced as far back as early Native American inhabitance (Greenlee and Langenheim 1990; Keeley 2002) and is a current management practice across a variety of habitat types (Carle 2008). Prescribed fire may aid in fuel reduction, lower the presence of invasive species (Pyne 1997), and suppress the encroachment of woody plants into grassland habitats (McBride and Heady 1968; Gordon 1995). However, to use fire as a tool for land management and conservation, we must understand the effects of prescribed burning on more than just the plant community.

Recent documentation of reptile declines (Reading et al. 2010; Sinervo et al. 2010) has made it apparent that population trends must be monitored and the consequences of land management practices on these species assessed (Gibbons et al. 2000). Fire may cause direct mortality or affect reptiles indirectly through habitat modification. Direct mortality of reptiles by fire has been found to be negligible in many studies (Howard et al. 1959; Kahn 1960; Vogl 1973; Floyd et al. 2002). In habitats historically maintained by fire, reptile species are assumed to be somewhat adapted to avoid direct mortality from fire (Means and Campbell 1981). Reptiles are thought to avoid harm by seeking shelter under debris or in burrows (Russell et al. 1999; Rochester et al. 2010, Pianka and Goodyear 2012).

Literature on the indirect effects of fire on reptiles demonstrates a variety of results among species (Greenberg and Waldrop 2008; Pianka and Goodyear 2012; Stephens et al. 2012) and with respect to habitat (Rochester et al. 2010; Valentine et al. 2012), season of fire (Hellgren et al. 2010), and management regime (Greenberg and Waldrop 2008). Changes in relative abundance are commonly species-specific rather than consistent across a community. Species-specific responses have been found in multiple habitat types including grasslands (Cavitt 2000; Setser and Cavitt 2003; Wilgers and Horne 2006; Ruthven et al. 2008), chaparral (Simovich 1979), mixed Mediterranean shrub and forest habitat (Santos and Poquet 2010), tropical forests (Braithwaite 1987), and tropical savanna (Faria et al. 2004; Valentine and Schwarzkopf 2009). Habitat preference, life history, and diet can influence how animals react to vegetation succession post-fire (Santos and Poquet 2010). Fire can cause change in habitat that affects predation (Wilgers and Horne 2007), resources

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(Hellgren et al. 2010; Pianka and Goodyear 2012), behavior (Lillywhite and North 1974; Hellgren et al. 2010), and thermal environment (Hossack and Corn 2007). Because response to fire depends on species, habitat type, and fire regime, more research on a variety of reptile species across different vegetation communities is needed to make effective and informed decisions on the management and conservation of reptiles.

Past research on the effects of fire on *Coluber constrictor* presents a clear example of the range of results obtained with a species that inhabits a variety of habitats (McLeod and Gates 1998; Cavitt 2000; Setser and Cavitt 2003; Wilgers and Horne 2007; Rochester et al. 2010). The polytypic species *C. constrictor* ranges from Guatemala to southern Canada and from the Pacific coast to the Atlantic coast; subspecies *C. c. mormon* is found west of the continental divide (Stebbins 2003). This subspecies is generally associated with open riparian, grassland, woodland, and chaparral habitats near water sources (Stebbins 2003; Lemm 2006; Stebbins and McGinnis 2012) and feeds primarily on insects but can also feed on small mammals, amphibians, and other reptiles (Shewchuk and Austin 2001). Several studies in tallgrass prairie indicate that immediately following fire, *C. constrictor* is adversely affected but that adverse effects diminish within one year post fire (Cavitt 2000; Setser and Cavitt 2003; Wilgers and Horne 2007). Long term unburned tallgrass prairie habitat may also be avoided by *C. constrictor* (Wilgers and Horne 2006). In pine forests, more *C. constrictor* were found in burned versus unburned forests (McLeod and Gates 1998). Rochester et al. (2010) found a decrease in *C. constrictor* occupancy of chaparral and coastal sage scrub habitats but not in grassland or woodland/riparian habitats two years post wildfire when compared to pre wildfire occupancy. The disparity in the response of *C. constrictor* in different habitats may reflect variation in the changes of the thermal properties, predation rates, and prey abundance caused by reduced vegetation cover in the respective habitat types (McLeod and Gates 1998; Wilgers and Horne 2007).

The objective of our study was to determine the effects of prescribed fire on *Coluber constrictor mormon* at a study site in coastal San Mateo County, California. We estimated the apparent survival, relative abundance, and body condition index of *C. c. mormon* in the burned and unburned sites of our study area. We also estimated percent cover of the main vegetation types in the burned and unburned sites. This research serves as a baseline for investigating the effects of fire on this species for future burns at our study site and adds to the knowledge base of the effects of fire on reptiles.

Materials and Methods

Study site.—Our study site consisted of an approximately 213-ha area in coastal southern San Mateo County, California (Figs. 1 and 2). The site contains several ponds and wetlands surrounded by upland habitat dominated by grassland and northern coastal scrub. Patches of riparian woodland dominated by willow (*Salix* spp.) and conifer forest dominated by Douglas-fir (*Pseudotsuga menziesii*) also occur within the study site. Elevation ranges from 5–190 m above sea level. Historically, livestock grazing occurred on this property; however, this practice ended in 1998. The study site is located on a property containing one of the largest remaining tracts of California coastal prairie (Peninsula Open Space Trust. 2012. Prescribed Fire. Available from http://www.openspacetrust.org/lands/stewardship_fire.html [Accessed 27 June 2012]). To

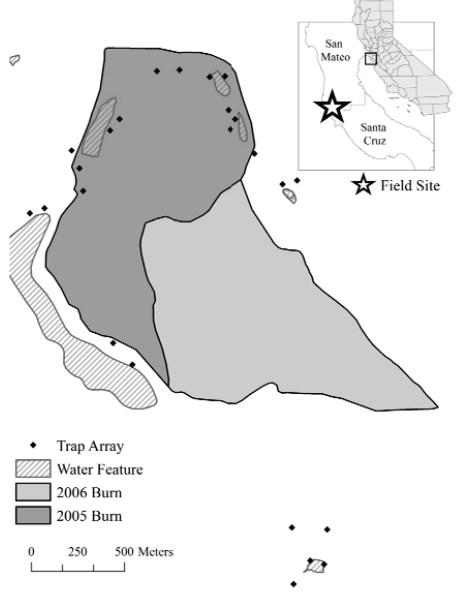

Figure. 1. Study site in coastal southern San Mateo County, California, 2007 to 2009 with trap locations.

FIGURE. 2. Photograph taken in 2010 of 2005 burn area showing riparian, grassland, and shrub and woodland components. Heterogeneity of vegetation types was high in both burned and unburned areas. (Photographed by Jeffrey J. Smith).

restore and maintain coastal prairie grasslands, a habitat historically maintained by fire (Ford and Hayes 2007), prescribed burns were conducted in the fall of 2005 (64 ha) and 2006 (67 ha) on the eastern portion of the property (Fig. 1). There is no previous documentation of prescribed fire or evidence of previous wildfires on this property (Jeff Powers, pers. comm.), but mean fire interval for the past century is estimated to be 20–30 years for coastal prairie habitat in this area (Greenlee and Langenheim 1990). Most of our surveys were focused in the 2005 burned site and adjacent unburned site (Fig. 1). Very few individuals were caught in the 2006 burn site; therefore, these individuals were excluded from the analysis. The area surrounding the 2005 and 2006 burn sites will hereinafter be described as the unburned site and the 2005 burned area will be described as the burned site. In 2007 through 2009, prescribed burns were planned but not executed for various reasons.

Snake sampling.—We used a combination of techniques (trap arrays, hand capture, and plywood and corrugated tin cover objects) to sample *C. c. mormon*. Trap arrays consisted of four funnel traps with two traps placed at each end of a 15-m drift fence. We placed traps approximately 3–215 m from ponds, wetlands, and reservoirs in each vegetation type (grassland, scrub, wetland, riparian forest, conifer forest), for a total of 13 trap arrays in the unburned site and 11 trap arrays in the burned area. Traps were placed in relative proportion to availability of each vegetation type and at varying distance from ponds to sample across the different types of vegetation on this property. We checked traps twice daily (morning and evening) for 277 trap days in June and July of 2007, 942 trap days March through June 2008 (10 days open, 14 days closed), 1,140 trap days April through August of 2009 (10 days open, 20 days closed), and 1,090 trap days May through June of 2010 (traps open continuously) resulting in a total of 3,449 trap days. Trapping in 2007 was sporadic because of permit delays. We extended surveys through August in 2009 to determine seasonal activity levels. Because this was the only year that we extended sampling past July and so few *C. c. mormon* individuals were captured (n = 8), we only used data from March through July in analyses (3,209 trap days). When possible, we captured additional *C. c. mormon* during visual encounter surveys, while checking traps and looking beneath cover objects (distributed in transects separate from trap arrays). Trap locations remained constant across years but the location of some cover object transects changed. We measured snout-vent length (SVL) and tail length, determined sex, weighed, and uniquely marked captured individuals with a microbrand (Winne et al. 2006) or Passive Integrated Transponder (PIT) tag. We released individuals at their location of capture within approximately 24 h of capture.

Vegetation sampling.—We surveyed vegetation in 2009 based on protocols set by Ralph et al. (1993). A 50 m radius plot surrounding each trap array was divided into three mutually exclusive percent coverage groups: riparian, non riparian, and road. We divided riparian vegetation into two layers: wetland vegetative coverage and pond water coverage, and non-riparian vegetation into five structure layers: tree, shrub, herbaceous cover less than one meter, herbaceous cover greater than or equal to one meter, and bare ground. For tree and shrub layers, we visually estimated dominant species coverage (species covering at least 10%). Average vegetation cover was summed by vegetation type (wetland, scrub/grassland, riparian woodland, conifer forest).

Mark-recapture analysis.—We were interested in multiple potential responses of *C. c. mormon* to fire, including the probability of survival, probability of transition (movement between burned and unburned sites), and abundance in the burned and unburned sites. Multi-state models allow for the estimation of population parameters in situations where animals change states stochastically (Lebreton et al. 1999; Lebreton and Pradel 2002), and were a natural choice for our study. In these models, probability of transition is defined as the probability of an individual moving from state A to state B at time $i + 1$, given that the animal was found in state A at time i.

We pooled trap array data by year from 2007 through 2010, for a total of four trapping occasions with site of capture (burned or unburned) as states. We excluded young of the year from analysis. Peak egg laying for *C. c. mormon* typically occurs June through July (Brown and Parker 1984; Rosen 1991) with an incubation period of approximately 41 to 44 d (Brown and Parker 1984), and neonates measure 225–261 mm SVL (Brown and Parker 1984; Rosen 1991). For the purpose of our analysis, we assumed that individuals caught on or after 1 June that measured < 262 mm SVL to be young of the year. We caught 12 of 669 *C. c. mormon* on both burned and unburned sites within the same year. Because data were pooled by year, the multiple states within a year could not be expressed in the analysis, and we designated the site at first capture of the year the state in the analysis.

Our main objectives for this analysis were to determine: (1) if apparent survival differed in the burned and unburned sites; (2) if the relationship of apparent survival and time differed between burned and unburned sites; (3) if there was a difference in probability of transition between the sites; and (4) if abundance differed between the burned and unburned sites (because of differences in survival and/or probability of transition). We also expected that there would be a difference in recapture rate between the sites because there was a different number of traps in each site and because snake behavior could have differed among burned and unburned areas.

We started with the global model Φ(site*t)p(site*t)ψ(site*t), with apparent survival (Φ), recapture rate (p), and probability of transition (ψ) dependent on site, time (year), and the interaction of site and time (where * indicates the interaction of effects) using the "multi-state recaptures only" formulation in Program MARK version 6.10 (White and Burnham 1999). We then evaluated simplifications of the global model to determine the best possible general model (from which to identify the candidate model set) that included the parameters necessary to address our objectives. Therefore, we tested for support of time dependence on parameters recapture rate and probability of transition. We did this by comparing the fit of our global model to the simplified models Φ(site*t)p(site)ψ(site*t), (site*t)p(site*t)ψ(site), and (site*t)p(site)ψ(site). Lastly, we evaluated all possible subsets of our best general model. All models were compared using Akaike's Information Criterion adjusted for small sample size (AIC$_C$). To test for goodness of fit, we used the median c-hat method in Program MARK with 1,000 bootstrap samples implemented on the global model.

We used the best-fit model as determined by AIC$_C$ to construct a multi-state Jolly-Seber model because of difficulties associated with model selection in a Bayesian analytical framework under data augmentation. This was appropriate because, although the model structure was similar, we are not aware of a multi-state Jolly-Seber model readily accessible in a frequentist analytical framework. Our model was based upon a generalization of the multi-state formulation of the Jolly-Seber model in Kéry and Schaub (2012) to allow recruitment into multiple states, state-specific transition probabilities, and state-specific survival probabilities. We used data augmentation under parameter expansion (Dupuis and Schwarz 2007; Royle 2009) to provide estimates of annual recruitment, the proportion of individuals recruited into each state, annual abundance of individuals in each state, and the total number of individuals ever alive at our site during the study. The augmented dataset consisted of 5,000 pseudo-individuals with all-zero capture histories. We analyzed the model using Markov chain Monte Carlo (MCMC) methods as implemented in JAGS 3.3.0 (Plummer 2003), called from R 2.15.2 (R Core Team 2012) using the package rjags (Plummer 2012). We ran the model on three chains of 10,000 iterations each, after a burn-in of 10,000 iterations and thinned by a factor of three for posterior inference based upon 10,000 samples from the posterior distribution. No evidence for lack of convergence existed, and the upper credible limit for superpopulation abundance was well below 5,000, indicating that a sufficient number of pseudo-individuals were used in the analysis.

Body condition.—To calculate body condition index (BCI), we used a regression analysis of log-transformed mass and log-transformed SVL in interaction with sex to calculate residuals of measurements for each snake (Madsen and Shine 2001; Lyet et al. 2009). Residuals were divided by predicted mass values to obtain the BCI, which represents the departure from the average mass for a given body length (Blouin-Demers et al. 2005; Coates et al. 2009). We examined the distribution of BCI data for normality. We included snakes captured using all methods in the analysis. To estimate body reserves without bias from clutch size or current stomach contents, we omitted gravid females, young of the year (as estimated by SVL), and individuals with large discernible gut contents. For individuals captured more than once, we randomly selected only one BCI value for this analysis. We then analyzed BCIs with a 2-way analysis of variance (ANOVA) to test for main effects of site (burned and unburned), year, and an interaction between site and year ($\alpha = 0.05$).

RESULTS

We captured 692 individuals 1,155 times March through July, 2007 through 2010. We had 134 captures in 2007 (72 unburned, 62 burned), 397 captures in 2008 (288 unburned, 109 burned), 279 captures in 2009 (171 unburned, 108 burned), and 345 captures in 2010 (214 unburned, 131 burned).

Mark-recapture analysis.—Our goodness of fit test resulted in a c-hat estimate of 1.16. Because our estimate was very close to one, we did not use c-hat to correct for overdispersion. Results from the evaluation of the effect of time dependence on our global model (Φ(site*t)p(site*t)ψ(site*t)) revealed no support for time dependence on probability of transition or recapture, resulting in a best general model (from which we identified our candidate model set) of Φ(site*t)p(site)ψ(site; Table 1A). Our best-supported model was Φ(t)p(.)ψ(site), with time dependence for apparent survival, constant recapture rate, and probability of transition dependent on site. However, considerable model selection uncertainty existed (Table 1B). We used the best-supported model as the structure for our multi-state Jolly-Seber model (γ(t*site)Φ(t)p(.)ψ(site), where γ = recruitment). Annual probability of survival differed by year but not site, and estimates ranged from 0.46 to 0.67, with the lowest estimate from the time period between 2008 and 2009 ($\Phi_{2007–2008} = 0.67$ [95% credible interval = 0.49–0.87], $\Phi_{2008–2009} = 0.46$ [0.36–0.58], $\Phi_{2009–2010} = 0.70$ [0.54–0.89]; Fig. 3). Mean abundance per trap array appeared to be fairly stable across years (Fig. 4). The unburned site had greater abundance than the burned site in all years, with a greater difference in abundance between unburned and burned sites in 2008 (23.7 [16.6–30.5] individuals per array) than in 2009 (9.7 [5.8–14.5] individuals per array) and 2010 (10.8 [6.3–16.4] individuals per array). We also found evidence for an effect of site on probability of transition (unburned to burned 0.06 [0.03–0.11]), burned to unburned 0.87 [0.79–0.94]). Annual individual capture probability was 0.42 (0.33–0.53).

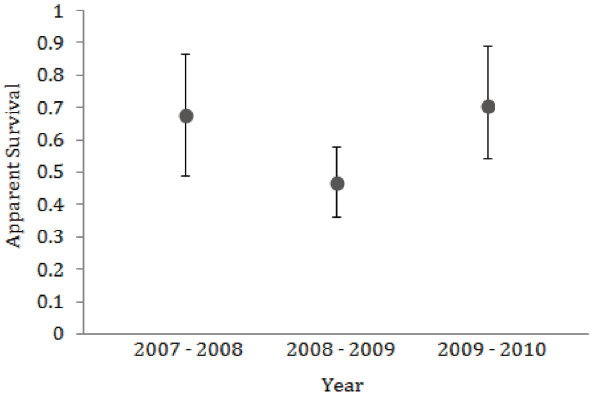

FIGURE 3. Annual apparent survival estimates for *Coluber coluber mormon* 2007 to 2010. Error bars represent 95% credible intervals.

Body condition.—BCI did not differ between sites ($F_{1,625} = 0.86$, $P = 0.35$), nor was there a significant site by year interaction ($F_{3,625} = 0.62$, $P = 0.61$). However, there was a significant year effect ($F_{3,625} = 7.20$, $P < 0.001$). BCI estimates for 2009 and 2010 were below average for both burned and unburned sites over the time period sampled (Fig. 5).

Vegetation.—Vegetation was similar between burned and unburned areas (Table 2). Both sites shared three shrub species with the most dominant cover (*Baccharis pilularis, Rubus* sp., *Toxicodendron diversilobum*) and two tree cover

Table 1. A) Model selection results for preliminary evaluation of the global model. B) Model selection results for all submodels of our best-supported general model from A). Only the top 14 models, which represent > 95% of the cumulative AICc weight, are included in the table. Parameters: Φ = apparent survival, p = probability of recapture, and ψ = probability of transition. A parameter that is assumed to be constant is represented as (.) and time dependence is represented as (t). The (*) denotes an interaction and (+) represents an additive effect. AICc = Akaike's Information Criterion corrected for small sample size, Model Likelihood = ratio of AICc weight from model in row over AICc weight of top model in set, n = 669.

Model	AICc	Δ AICc	AICc weight	Model Likelihood	No. par.	Deviance
A)						
Φ(site*t)p(site)ψ(site)	1003.80	0.00	0.83	1.000	10	52.89
Φ(site*t)p(site*t)ψ(site)	1007.35	3.54	0.14	0.17	12	52.28
Φ(site*t)p(site)ψ(site*t)	1011.21	7.40	0.02	0.02	14	51.96
Φ(site*t)p(site*t)ψ(site*t)	1014.90	11.09	0.00	0.00	16	51.43
B)						
Φ(t)p(.)ψ(site)	1001.17	0.00	0.16	1.00	6	58.48
Φ(t)p(.)ψ(.)	1001.33	0.16	0.15	0.92	5	60.68
Φ(t)p(site)ψ(site)	1001.67	0.51	0.13	0.78	7	56.94
Φ(site + t)p(site)ψ(site)	1001.77	0.62	0.12	0.73	8	55.00
Φ(site + t)p(.)ψ(site)	1003.21	2.04	0.06	0.36	7	58.48
Φ(t)p(site)ψ(.)	1003.26	2.09	0.06	0.35	6	60.58
Φ(site + t)p(.)ψ(.)	1003.34	2.17	0.06	0.34	6	60.66
Φ(site*t)p(site)ψ(site)	1003.80	2.64	0.04	0.27	10	52.89
Φ(site)p(.)ψ(.)	1003.91	2.75	0.04	0.25	6	61.23
Φ(.)p(site)ψ(site)	1004.12	2.95	0.04	0.23	5	63.48
Φ(.)p(.)ψ(site)	1004.78	3.61	0.03	0.16	4	66.17
Φ(.)p(.)ψ(.)	1004.95	3.78	0.03	0.15	3	68.37
Φ(site + t)p(site)ψ(.)	1005.02	3.86	0.02	0.15	7	60.29
Φ(site*t)p(.)ψ(site)	1005.99	4.82	0.01	0.09	9	57.15

species (*Pseudotsuga menziesii* and *Salix* sp.). In scrub/grassland vegetation, burned and unburned sample plots had similar vegetation percentage cover, except the burned site had a small percentage of dead trees with evidence of damage by fire. In wetland vegetation, average percentage shrub coverage at the burned site was about half that of the unburned site. Percentage tree cover was higher in burned riparian woodland and conifer forest; however, very few plots were measured for these habitat types (Table 2).

Discussion

This study provides information on the survival, relative abundance, and body condition of *C. c. mormon* on both burned and unburned areas of our study site spring and summer of 2007 through 2010. Overall, it appears that the fires conducted on this property did not have a major effect on *C. c. mormon* two to five years post-fire. We found no difference in apparent survival or body condition of snakes between the burned and unburned sites, and abundances in 2009 and 2010 were similar. However, we did

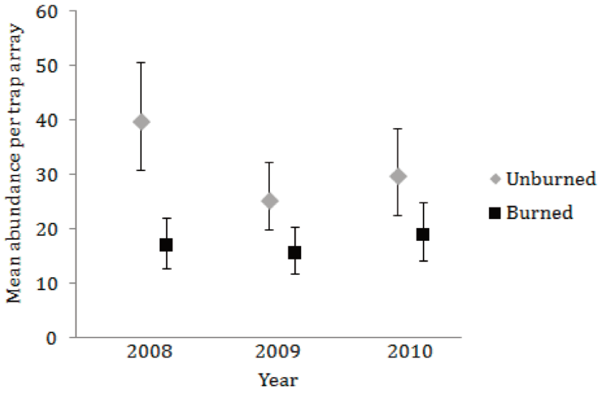

FIGURE 4. Annual mean relative abundance (abundance per trap array) of *C. c. mormon* in burned and unburned sites 2008 to 2010. Error bars represent 95% credible intervals.

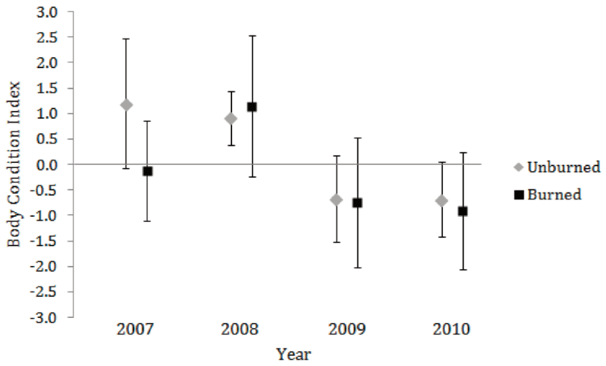

FIGURE 5. Annual average body condition estimates of *C. c. mormon* in burned and unburned sites 2007 to 2010. Error bars represent approximate 95% confidence intervals. $n_{unburned} = 420$ ($n_{2007} = 41$, $n_{2008} = 163$, $n_{2009} = 108$, $n_{2010} = 108$), $n_{burned} = 213$ ($n_{2007} = 35$, $n_{2008} = 57$, $n_{2009} = 53$, $n_{2010} = 68$).

find evidence for a higher snake abundance in the unburned site in 2008 and higher rate of movement of snakes from the burned site to the unburned site.

Although more research is needed to confirm the long and short term effects of fire on this species, there may be several reasons why the prescribed burns conducted at this site could have had a negligible impact on *C. c. mormon.* Herpetofaunal abundance and diversity are closely tied to vegetation type and structure (Means and Campbell 1981; Block and Morrison 1998; Valentine and Schwarzkopf 2009). Removal of vegetative cover by fire can be beneficial to species that prefer a more open habitat that confers thermoregulatory advantages (Landers and Speake 1980; Mushinsky 1985; Hossack and Corn 2007; Ruthven et al. 2008), or can have a negative effect on populations if species prefer substantial cover such as leaf litter (McLeod and Gates 1998; Moseley et al. 2003; Schurbon and Fauth 2003; Valentine and Schwarzkopf 2009) or if reduced vegetation cover exposes reptiles to an increase in predation (Wilgers and Horne 2007). Reduced vegetation cover has been implicated as a major factor in *C. constrictor* response to fire because of increased vulnerability to predators in grasslands (Setser and Cavitt 2003; Wilgers and Horne 2007) and change in thermal or moisture conditions (McLeod and Gates 1998; Setser and Cavitt 2003). The prescribed burns on our study site were low to medium intensity fires with sporadic high intensity fire areas that resulted in limited penetration into pond and wetland habitat or conifer and dense shrub stands (Jeff Powers, pers. comm.). We found little difference in shrub cover between the burned and unburned sites, and grass cover has been observed to return within one growing season after fire (Jeff Powers, pers. comm.). Therefore, it is not surprising to find little difference in *C. c. mormon* survival, BCI, and abundance between sites two to five years after fire. Furthermore, the burned sites were surrounded by unburned habitat with similar vegetative cover and pond and wetland features. Even if grassland within the burned area was significantly reduced in cover immediately after the fire, adjacent unburned sites could have provided refugia for animals until cover at the burned site returned.

It is possible that *C. c. mormon* abundance at our study site was initially considerably reduced post fire but returned with the growth of herbaceous cover by the time we initiated our study two to three years after burning. Our results indicate there may have been a successive decline in the negative effect of the burned site on abundance, although estimates were of too short a duration to confirm this trend. A successive decline in adverse effect would be consistent with other studies on grassland fires. *Coluber constrictor* abundance decreased after spring fire in tallgrass prairie habitat (Cavitt 2000; Setser and Cavitt 2003); however, abundances were similar between burned and

Table 2. Average percentage vegetation cover ± 1 SD in four different vegetation types at burned and unburned sites, measured in 2009.

	Wetland		Scrub/grassland		Riparian woodland		Conifer forest	
Habitat variable	Unburned	Burned	Unburned	Burned	Unburned	Burned	Unburned	Burned
Number of sample plots	3	3	5	3	1	1	2	1
Tree % cover	11.67 ± 12.58	5.00 ± 8.66	10.00 ± 13.69	11.67 ± 10.41	15.00	40.00	17.50 ± 3.54	50.00
Dominant tree species % cover								
Cornus occidentalis	1.00 ± 1.73	0.00 ± 0.00	0.00 ± 0.00	0.75 ± 1.30	0.00	4.00	0.00 ± 0.00	0.00
Pseudotsuga mensiesii	2.33 ± 4.04	0.00 ± 0.00	5.00 ± 11.18	4.67 ± 8.08	0.00	0.00	17.50 ± 3.54	42.50
Pseudotsuga mensiesii (dead)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	2.00 ± 3.46	0.00	0.00	0.00 ± 0.00	7.50
Myrica californica	0.00 ± 0.00	2.50 ± 4.33	0.00 ± 0.00	3.25 ± 5.63	0.00	8.00	0.00 ± 0.00	0.00
Salix sp.	7.92 ± 13.71	1.50 ± 2.60	5.00 ± 11.18	1.00 ± 1.73	15.00	28.00	0.00 ± 0.00	0.00
Shrub % cover	45.00 ± 5.00	23.33 ± 7.64	41.00 ± 26.08	40.00 ± 0.00	40.00	40.00	52.00 ± 10.61	30.00
Dominant shrub species % cover								
Baccharis pilularis	14.00 ± 1.80	8.58 ± 4.30	17.25 ± 10.31	12.00 ± 6.93	14.00	8.00	13.12 ± 2.65	9.00
Heteromeles arbutifolia	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00	0.00	3.00 ± 4.24	0.00
Lonicera involucrata	1.33 ± 2.31	0.00 ± 0.00	0.00 ± 0.00	2.00 ± 3.46	0.00	4.00	0.00 ± 0.00	0.00
Rhamnus californica	2.50 ± 4.33	0.50 ± 0.87	0.00 ± 0.00	1.33 ± 2.31	0.00	6.00	6.75 ± 2.83	0.00
Rubus sp.	16.33 ± 1.53	8.08 ± 2.50	11.90 ± 12.86	14.00 ± 5.29	14.00	10.00	16.50 ± 2.12	13.50
Toxicodendron diversilobum	8.00 ± 7.55	4.58 ± 2.60	7.60 ± 6.35	8.00 ± 8.00	10.00	8.00	9.75 ± 7.42	6.00
Herbaceous % cover < 1 m	38.33 ± 20.21	46.67 ± 2.89	62.00 ± 26.83	43.33 ± 15.28	20.00	25.00	27.50 ± 10.61	20.00
Herbaceous % cover ≥ 1 m	10.00 ± 5.00	13.33 ± 2.89	14.00 ± 8.22	26.67 ± 15.28	10.00	5.00	20.00 ± 7.07	30.00
Bare ground % cover	0.00 ± 0.00	0.00 ± 0.00	1.00 ± 2.24	0.00 ± 0.00	0.00	0.00	0.00 ± 0.00	30.00
Wetland % cover	25.00 ± 8.66	33.33 ± 7.64	3.00 ± 4.47	3.33 ± 5.77	30.00	30.00	0.00 ± 0.00	0.00
Pond % cover	31.67 ± 5.77	31.67 ± 10.41	2.00 ± 4.47	0.00 ± 0.00	40.00	30.00	0.00 ± 0.00	0.00
Road % cover	3.33 ± 2.89	0.00 ± 0.00	4.00 ± 4.18	3.33 ± 5.77	0.00	0.00	5.00 ± 7.07	0.00

unburned plots in fall of the same year (Setser and Cavitt 2003). Wilgers and Horne (2007) found that predation risk on *C. constrictor* was higher in recently burned tallgrass prairie one to two months post fire, and activity level of snakes within burned plots was reduced only up to 20 d post fire. Rochester et al. (2010) found reduced occupancy in coastal sage scrub and chaparral sites with drastically reduced cover after a large-scale wildfire; however, there was no significant change in *C. constrictor* occupancy in grassland and woodland/riparian plots where shrub and tree cover was not significantly reduced by fire.

We found vegetation cover in the burned and unburned sites to be similar in 2009. The three dominant shrub species at this site were *Baccharis pilularis, Rubus* sp., and *Toxicodendron diversilobum. Rubus* sp. and *Toxicodendron diversilobum* may be top-killed from fire but commonly resprout after disturbance (Keeley 2006), and fire may be good at reducing *Baccharis* spread but not effective at eliminating existing dense shrub stands (McBride 1974). Although these prescribed fires may not dramatically reduce shrub cover, continued use of fire may hinder the spread of shrubs in the burned area, maintaining a heterogeneous mix of open grassland and more closed shrub and tree-dominated patches. Just as reduced cover can have an impact on reptiles, too much vegetation cover can also have negative impacts (Ballinger and Watts 1995; Webb et al. 2005). If fire can be used to maintain a variety of vegetation types, then prescribed fire may have long term benefits for this species. Long term data are needed to see if fire is effective at vegetation management at this site and if habitat differences as a result of fire have a significant effect on this species.

Our data support asymmetrical transition rates, with more individuals moving off of the burn site onto the unburned site, although the underlying cause of this asymmetry is unclear. Home range estimates for *C. constrictor* vary greatly by location (0.4–12.2 ha, minimum convex polygon; Fitch and Shirer 1971; Brown and Parker 1976; Plummer and Congdon 1994; Klug et al. 2011). However, if the *C c. mormon* at our study site fall anywhere within these previously reported values, it is probable that a number of snakes at our study site were moving between parts of the burned and unburned areas within their home range prior to application of fire. Trap arrays were located 12–810 m from the boundary between burned and unburned sites and 11 of 24 trap arrays were within 100 m of the boundary. The relatively small burned area was surrounded by unburned habitat and some of the unburned trap arrays were much farther away from the burn boundary than any of the burned trap arrays, making transitions from the burned area to the unburned area *a priori* more likely (Fig. 1). Because we lack pre-fire data, we were not able to compare the probability of transition to pre-burn conditions. Therefore, we cannot discern if the asymmetry in the probability of transition is due to *C c. mormon* preferentially moving from one site to another or if the difference in transition rates is solely a reflection of our trapping scheme. Also, BCI was similar between burned and unburned sites but varied significantly among years, showing decreasing trends later in our study period. Apparent survival also dipped between 2008 and 2009, suggesting that *C. c. mormon* may have undergone some sort of negative pressure on both the burned and unburned sites.

The results of our study provide baseline information of the effects of fire on *C. c. mormon* at this site and show support that the past fires have not been detrimental to *C. c. mormon*. However, to gain more detailed information of the effects of fire on this species, analyses should be done with multiple burned and unburned sites. We used snake as the experimental unit in mark-recapture and BCI analysis within the one burned and one unburned area, limiting our conclusions to the direct comparison of the two areas sampled within our study site. Although the experimental design limits our conclusions, we find it unlikely that this study would show such low support for site difference if fire had a substantial effect on this population. If the prescribed burns were the cause of a considerable source of direct or indirect pressure on this species, we would expect to have found large survival, BCI, and abundance differences between sites, and our estimated apparent survival probabilities are within the range reported in other studies on *C. constrictor* (0.54–0.79; Brown and Parker 1984; Rosen 1991). Furthermore, there was no indication of an interaction of site with year for apparent survival or BCI, indicating that temporal changes were similar on both sites during our study.

It is important to monitor additional burns at this site for effects on *C. c. mormon*. Common species (such as *C. c. mormon* at our study site) can make up a large portion of the biomass and play a large role in trophic interactions and nutrient cycling, making them integral to our understanding of the effects of disturbance on an ecosystem (Gaston 2010). Common species are also useful for biological monitoring because larger sample sizes allow for the detection of smaller changes in a system. *Coluber constrictor* has a wide geographic range and inhabits many different habitat types. Understanding the effects of prescribed fire on this species has broad implications for land management decisions for areas containing *C. constrictor* and species with similar habitat use. This study site is unique because of the presence of California coastal prairie, a habitat that has been heavily impacted by human influence (Noss and Peters 1995). Therefore, continued study at this site allows for the opportunity to study the largely unknown relationship of this species with fire within this uncommon habitat. Long term data, preferably spanning multiple generations, are needed to differentiate natural population fluctuations in survivorship and abundance from effects of habitat alteration (Pechmann et al. 1991). It is also important to collect additional data on the other reptile and amphibian species that occur at this site. No one fire regime will be optimal for all flora and fauna. It is important to balance the needs of the whole community while taking special precaution for sensitive species.

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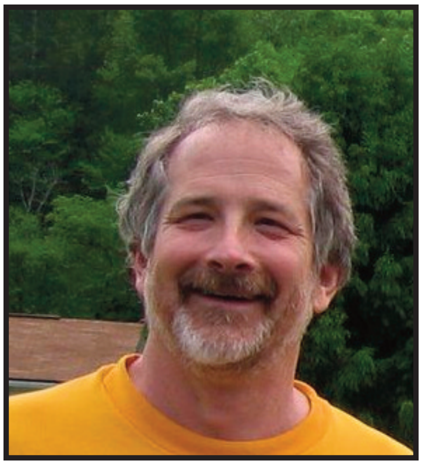

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