WESTERN WILDLIFE

THE JOURNAL OF THE WESTERN SECTION OF THE WILDLIFE SOCIETY

Photo: John H. Harris

VOLUME 6 • 2019

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Peer-Edited

Notes

Range Extension of the Western Yellow Bat (*Dasypterus xanthinus*) in New Mexico

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Abstract.—The Western Yellow Bat (*Dasypterus* [= *Lasiurus*] *xanthinus*), a state-listed threatened species in New Mexico, occurs in the southwestern U.S. and has been previously reported in New Mexico only from the southwestern corner of the state. We document its occurrence based on photographs of a specimen from Las Cruces, Doña Ana County, New Mexico, which represents a northeastward range extension from previous localities in the state. Our record also documents the use of fan palms (*Washingtonia* spp.) in New Mexico as day roost habitat. We discuss distribution and habitat use of *D. xanthinus* in the Southwest.

Key Words.—bats; Chihuahuan Desert; geographic distribution; fan palm; *Lasiurus xanthinus*; roost habitat; *Washingtonia*.

The Western Yellow Bat, *Dasypterus xanthinus*, is a tree-roosting vespertilionid species that ranges from the southwestern U.S. to southern Mexico (Simmons 2005; Harvey et al. 2011). The bat was formerly classified as a subspecies of *Lasiurus ega* (e.g., Kurta and Lehr 1995), but genetic studies have supported its recognition as a distinct species (Baker et al. 1988; Morales and Bickham 1995; Simmons 2005). More recently, Baird et al. (2015) proposed the reassignment of all yellow bat species from the genus *Lasiurus* to *Dasypterus*. In the U.S., *D. xanthinus* has been most often recorded from southern California and central and southern Arizona (Constantine 1966, 1998; Hoffmeister 1986) and was more recently documented in southern Nevada (O'Farrell et al. 2004). The species was first discovered in Texas in 1996 and is currently known from multiple sites in the western part of that state (Higginbotham et al. 1999; Ammerman et al. 2012; Schmidly and Bradley 2016). The geographic range in New Mexico has been considered to be peripheral and limited to the southwestern corner of the state (Findley et al. 1975; Cook 1986). Herein we document a new locality record and range extension in New Mexico, which partially fills a distributional gap in the southern part of the state.

On 2 January 2019, four *D. xanthinus* (unknown sex) were discovered roosting in the canopy of a mature ornamental fan palm tree (*Washingtonia filifera* × *robusta*; Fig. 1) by two workers (Kailin Miller and Jesus Rodriguez) while they were pruning the palms in a city park in Las Cruces, Doña Ana County, New Mexico (32.34586, -106.77520, WGS 84; 1,234 m elevation).

Figure 1. Ornamental fan palm trees (*Washingtonia filifera* × *robusta*) where Western Yellow Bats (*Dasypterus xanthinus*) were found roosting in a city park in Las Cruces, New Mexico, 2 January 2019. The four bats were found in the dead fronds of the fourth tree from the left. (Photographed by Jesus Rodriguez).

The bats, which appeared to be torpid, were observed sequestered within the cluster of dead fronds encircling the trunk below the live fronds as the dead growth was being trimmed and removed (Fig. 2). After their removal from the tree, the bats were briefly lethargic, thereby allowing photographs of one individual (Fig. 3), before they became active and flew off. Ambient temperature on this date in Las Cruces ranged from about -4 to 14° C. The photographs were sent to us and we identified the photographed individual as *D. xanthinus* based on the pale yellow color of the pelage, the presence of fur

FIGURE 2. A Western Yellow Bat (*Dasypterus xanthinus*) found roosting among dead fronds of a fan palm tree at Las Cruces, New Mexico, 2 January 2019. (Photographed by Kailin Miller).

FIGURE 3. A Western Yellow Bat (*Dasypterus xanthinus*) following removal from dead fronds of a fan palm tree at Las Cruces, New Mexico, 2 January 2019. (Photographed by Kailin Miller).

on the anterior part of the uropatagium, the blackish color of the wing membranes, the lack of a dark face mask, and an estimated total body length of 105 mm. This combination of characteristics (after Ammerman et al. 2012) are consistent with only this species in New Mexico. The other three bats found in the same tree were not photographed but had similar characteristics and were presumably also *D. xanthinus*.

Prior to its discovery in Las Cruces, *D. xanthinus* was known in New Mexico only from southern Hidalgo County, in the southwestern corner of the state, where it was first detected in 1962 in Guadalupe Canyon in association with a riparian woodland of Eastern Cottonwood (*Populus deltoides*), Arizona Sycamore (*Platanus wrightii*), and Netleaf Hackberry (*Celtis reticulata*; Mumford and Zimmerman 1963). Additional specimens were mist-netted in the early 1980s at stream pools and livestock ponds near the Animas Mountains (Cook 1986). All specimens from New Mexico have been collected during late spring or summer (Cook 1986). Due to its very restricted range and presumed reliance on limited riparian woodlands in New Mexico, *D. xanthinus* has been listed by the state as threatened under the New Mexico Wildlife Conservation Act since 1975 (Jones and Schmitt 1997).

The Las Cruces record provides a northeastward range extension in New Mexico of about 210 km and is the first New Mexico record from the Chihuahuan Desert ecoregion. The species, however, has been verified to occur in adjacent El Paso County, Texas (about 60 km to the southeast; Tipps et al. 2011), as well as elsewhere in the Chihuahuan Desert ecoregion of Trans-Pecos Texas. Its occurrence, therefore, in south-central New Mexico is not unexpected. Other researchers have suggested that *D. xanthinus* is expanding its range in the U.S., perhaps in response to the planting of ornamental trees such as fan palms that are suitable as day roost sites and possibly as winter refugia (e.g., Barbour and Davis 1969; Constantine 1998). Although the Las Cruces record might represent a recent range expansion, we cannot exclude the possibility that the species has simply evaded detection in this area until recently.

Our record also provides the first evidence of *D. xanthinus* using a fan palm tree (*Washingtonia filifera × robusta*; Fig. 1) as a day roost site in New Mexico and is the first winter record of this species in the state. The species is known to use the dead fronds of fan palms as both summer and winter roost sites in Arizona, southern Nevada, and southern California (Hoffmeister 1986; Constantine 1998; O'Farrell et al. 2004), where native and non-native palms are more common, particularly as ornamentals. In New Mexico, suitable climate for non-native ornamental palms is limited to a few southern locations such as Las Cruces, Truth or Consequences, Carlsbad, and Deming. In western Texas, *D. xanthinus* has been found roosting in the dead fronds of Giant Dagger (*Yucca carnerosana*; syn. *Y. faxoniana*), a plant with structural similarities to fan palms that apparently make it a suitable shelter (Higginbotham et al. 2000). Yucca plants (*Yucca* spp.) are common in much of the Chihuahuan Desert ecoregion of southern New Mexico, including in the Las Cruces area, which suggests that suitable roosting habitat for *D. xanthinus* might be more widespread and available in the state than previously believed.

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Survey of Potential Predators of the Endangered Amargosa Vole (*Microtus californicus scirpensis*)

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Abstract.—As part of a comprehensive program assessing threats to the persistence of the endangered Amargosa Voles (*Microtus californicus scirpensis*) in the Mojave Desert of California, we used point counts, owl call surveys, camera-trapping, and scat transects to investigate diversity and activity of potential predators near Tecopa, California, USA. Of 31 predator species within the critical habitat of the vole, the most commonly detected were Coyotes (*Canis latrans*), Domestic Dogs (*C. lupus familiaris*), and Great Blue Herons (*Ardea herodias*). Predator species richness and detections were highest in the northern part of the study site where voles are more abundant. Predator detections were most common in the fall. We observed vole remains in 3.9 % of scat or pellet samples from Coyotes, Bobcats (*Lynx rufus*), and Great-horned Owls (*Bubo virginianus*). These data can support management activities and provide needed baseline information for assessment of the impact of predators on Amargosa Voles, including whether over-predation is limiting recovery and whether predators regulate this species.

Key Words.—endangered species; Mojave Desert; population regulation; species richness.

Introduction

Predators can regulate prey populations (Korpimaki et al. 2002; Banks et al. 2004), limit prey dispersal and patch colonization (Nie and Liu 2005; Smith and Batzli 2006) and restrict prey species to sub-optimal patches (Fey et al. 2006; Eccard et al. 2008). Overharvest of endangered prey species by predators may reduce prey population viability (Hartt and Haefner 1995), particularly when predator numbers are maintained by common prey species that are sympatric with endangered prey. The Amargosa Vole (*Microtus californicus scirpensis*) is a Mojave Desert rodent that is federally listed as endangered and has one of the narrowest niche breadths of any North American mammal (U.S. Fish and Wildlife Service [USFWS] 1997). Limited, fragmented, and lost habitat, low genetic diversity, predation, and disease all impact Amargosa Vole persistence (USFWS 1997; Ott-Conn et al. 2014).

Fewer than 500 individuals exist, occupying approximately 36 marsh patches near Tecopa, Inyo County, California (Cudworth and Koprowski 2010; Janet Foley et al., unpubl. report). Survival rates as low as 0.35 individuals/month were inferred to be caused at least partly by predation (Klinger et al. 2015). With low survival rates and high variability in population growth rates, population viability analysis predicted unacceptably high risks of extinction within 20–24 y (Foley and Foley 2016). Importantly, impacts of predators on voles could be enhanced by abundance of sympatric prey species. The Recovery Plan for Amargosa Voles (USFWS 1997) lists the study of predation on Amargosa Voles as an important task. The primary objective of our research was to inventory potential predators of the Amargosa Vole by species and guild. We also explored data for temporal trends, geographical distributions, and habitat associations in predator detections.

Methods

Study site.—We conducted this study near Tecopa in the Mojave Desert in southeastern Inyo County, California. The climate is characterized by wide daily and annual fluctuations in temperature, from a mean winter low of 3.2° C to a mean summer high of 41.0° C (www.ncdc.noaa.gov). The region experiences low and variable precipitation with mean annual rainfall of 12.3 cm. Amargosa Voles are almost completely dependent on Olney's Threesquare Bulrush (*Schoenoplectus americanus*) for food and cover (Klinger et al. 2015). Additional common plant species include rushes (*Juncus* spp.), Common Reed (*Phragmites australis*), cattail (*Typha* sp.), Salt Grass (*Distichlis spicata*), Yerba Mansa (*Anemopsis californica*), Boraxweed (*Nitrophila occidentalis*), Slender Arrowgrass (*Triglochin concinna*), Alkali Sacaton (*Sporobolus airodes*), mesquite (*Prosopis* spp.), and other wetland and desert plants (Rado and Rowlands 1984).

Marshes inhabited by voles tend to be patchy, fed by springs and surface flow, and from 290–420 m in elevation (Janet Foley et al., unpubl. report). There are approximately 40 marsh patches in the Tecopa region and three in Shoshone. For this study, we surveyed in and near marshes with and without Amargosa Voles, in
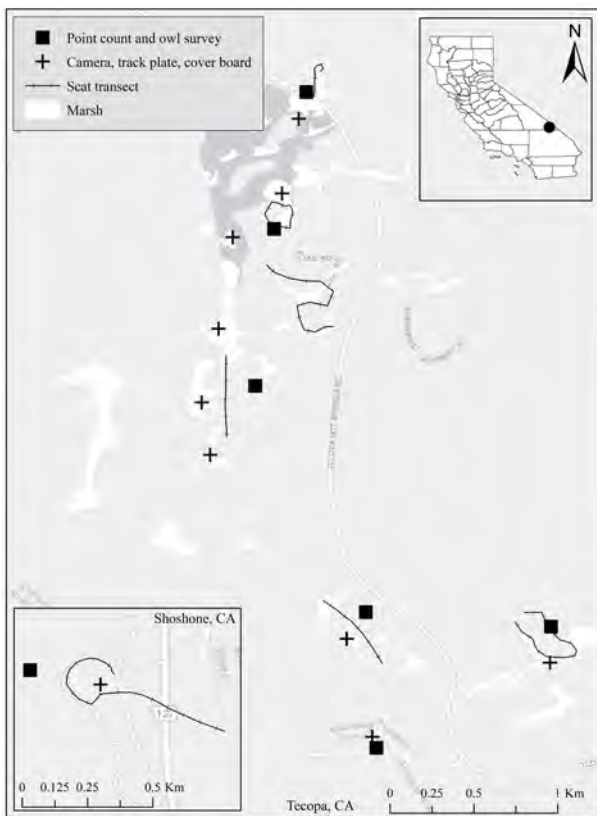
Figure 1. Locations near Tecopa and Shoshone, Inyo County, California, where surveys for predators of Amargosa Vole (*Microtus californicus scirpensis*) were conducted using point counts, owl call surveys, camera traps, and cover boards.

three different elevation zones (playa to the west, canyon south of the Old Spanish Trail, and in the Tecopa Hills on the east side of the site), and along an array from north to south (Shoshone, north Tecopa, south Tecopa separated at a natural break among marshes; Fig. 1). Each survey method (e.g., cameras, point counts, etc.) was conducted with a different spatial focus as described below. Sites with and without voles were included to evaluate whether voles might be an attractant for certain types of predators, or conversely, if certain predator assemblages affected vole presence. The elevation zones were chosen because of similarities within zone in vegetation community and hydrogeology. Playa areas tended to be very flat, and marshes in the playa were surrounded by alkaline playa dirt with very little vegetation cover. Hills housed the sources of most local warm springs and marshes in this zone were surrounded by moderately more diverse vegetation, while towards the canyon, the Amargosa River comes above ground, vegetation is progressively more diverse, and patches between marsh are characterized by saltbush scrub (Barbour et al. 2007).

Field methods.—We collected data on potential predators of voles between October 2013 and December 2014. We obtained data on vole presence from complementary live-trapping and sign survey studies (Deana Clifford et al., unpubl. report). Because our goal

was to identify as many different predator species in the range of Amargosa Voles as possible, we implemented camera-trapping, cover boards, point counts, callback surveys, and scat surveys, and recorded incidental observations. Stations for each of these survey techniques differed: (1) we performed callbacks and point counts on hilltops to maximize hearing and viewing of predators at multiple nearby marshes, (2) camera-trapping within marsh patches, and (3) scat surveys along inter-marsh transects (Fig. 1).

For camera trapping, we chose nine marshes that were evenly spaced over the range of Amargosa Voles and accessible from a road in Tecopa (Fig. 1), and a tenth marsh that was in the type locality for the Amargosa Vole in Shoshone, California, and was undergoing restoration for future vole reintroduction. At each site, we deployed 2-3 RC Covert or PC900 HyperFire (Reconyx, Holmen, Wisconsin, USA) cameras if a marsh was < 0.5 ha or 4–5 cameras in marshes > 0.5 ha, for a total of 36 cameras. Where signs of Amargosa Voles were absent at a marsh, we baited cameras with predator lure (Carmin's Canine Call, New Milford, Pennsylvania, USA, and Caven's Terminator Bait, Pennock, Minnesota, USA) to increase sensitivity, but we did not bait cameras in vole-occupied marshes so as not to increase predation on voles. We mounted cameras to U-posts using bailing wire and placed them along the marsh periphery near game trails or predator scat. We trimmed vegetation as needed to minimize false triggers, and programmed cameras to take five photographs at a time, with no delay between the next set of images if the camera was triggered. Cameras were active for 13 mo starting November 2013. We downloaded camera data monthly, although occasionally memory cards filled in less than a month. Skilled personnel (ADR, ANR) recorded date, time, and species of predators from images. Each of the 10 trapped marshes also received two 1-m^2 cover boards under which snakes and lizards were expected to hide (Grant et al. 1992), which we checked once per month.

We performed predator point counts at seven high vantage points from which we could see and hear predators in the same marshes assessed by cameras, as well as 30 additional marshes and playa between marshes. We surveyed the marshes and playa for diurnal and crepuscular species with binoculars one day per month, three times each day (dawn, mid-day, and dusk) for 15 min. At these same point count locations, we also conducted monthly 15-min callback surveys beginning approximately one hour after dusk, using recorded calls of the three most common owl species (Leonard Warren, personal communication): Great-horned Owl (*Bubo virginianus*), Long-eared Owl (*Asio otus*), and Barn Owl (*Tyto alba*). Although calls could attract predators, the duration of the survey was very short and we thought it unlikely to represent a risk to voles.

Lastly, we conducted monthly scat surveys beneath roost sites in trees and power-lines and along seven

200–700 m long transects located proximally to camera-trapping sites (Fig. 1). We recorded numbers of scat, removing samples from transects to avoid duplicate observations in subsequent surveys. Owl pellets were tentatively attributed to species based on repeated observations of a single owl species at the roost site. We identified the species of scat using a field guide (Elbroch 2003). When rodent bones were present, we disinfected the material in mild bleach solution, and identified bones to the lowest taxonomic level possible (Lawlor 1979; Jones and Manning 1992).

Analysis. —We maintained data in Excel and analyzed them with the statistical program R (http://www.r-project.org). We used Sanderson's AllPictures Method to differentiate repeated camera shots of the same predator individual from differing individuals (Sanderson and Harris 2013). We inferred statistical significance at $P \leq 0.05$. We compiled a comprehensive list of vole predators from all assays and reported numbers of observations by point counts, camera-trapping, and incidental observation grouped into the following guilds: aerial hunters, pursuit hunters, waders, and non-native (Table 1). Summary statistics of point counts and camera data included species richness (S) and number of records by guild for north vs. south marshes, and whether the marsh was in the Amargosa Canyon, hills, or desert playa. We examined differences in S by region (hills, playa, or canyon) using ANOVA and by district (north or south) with a Student's *t*-test.

Because of bias due to some marshes having baited cameras, we did not perform spatial statistical analyses for predators. We did compile data from marshes with and without bait to perform temporal statistical summaries. We used a Mann-Whitney U test to assess whether baited and unbaited marshes differed in median detections per marsh. We evaluated whether the number of what we considered independent camera detections (i.e., occurring at least 120 min apart) of all predator species combined and of the most common species differed seasonally (winter: December-February, spring: March-May, summer: June-August, fall: October-November), and between night (between sunset and sunrise) and day (between sunrise and sunset) using Poisson regression. We calculated prevalence of vole remains in predator scat and pellets (number of scats or pellets with vole remains/ number total number of scats or pellets examined) using the prop.test function in R, and inferred statistical differences in prevalence among the three predator species whose scat or pellets contained vole bones using a Chi-square test.

Results

We recorded 31 predator species observed over the course of this study (Table 1). The most commonly detected were Common Ravens (*Corvus corax*),

Northern Harriers (*Circus cyaneus*), Great Blue Herons (*Ardea herodias*), Great Egrets (*Ardea alba*), Coyotes (*Canis latrans*), Bobcats (*Lynx rufus*), and Dogs (*Canis lupus familiaris*). Incidental observations of Long-eared Owls, Greater Roadrunners (*Geococcyx californianus*), California Kingsnakes (*Lampropeltis californiae*), Gopher Snakes (*Pituophis catenifer*), a Coachwhip (*Masticophis flagellum*), a Spotted Skunk (*Spilogale gracilis*), and Domestic Cats (*Felis catus*; Table 1) were made. Anecdotally outside the temporal scope of this study, a Desert Kit Fox (*Vulpes macrotis arsipus*) was observed on camera in a marsh occupied by voles. We checked cover boards once per month, resulting in 240 trap nights, but did not detect any predators under cover boards.

Cameras recorded 13,614 camera-trap days across 10 sample marshes, resulting in the detection of 15 predator species (Table 1), including Bobcats, which were not detected through other methods and were only detected at baited camera sites. Of 8,520 images that clearly showed a predator, there were 831 independent events. Overall camera-trap success (number of independent events/number of camera-trap days) was 5.5%. The most frequently observed species on camera was Coyote, occurring in all 10 sampled marshes and at least once within each sample period. Baited cameras (two marshes) yielded 282 total detections and unbaited cameras (eight marshes) yielded 549; however, the average number of detections across periods was significantly higher (1.54) in unbaited compared with baited (1.29) marshes ($t = 2.99$, df = 574, $P = 0.003$).

Overall detections were more common during the day than night ($X^2 = 15.37$, df = 1, $P < 0.001$), with all common species being more active in day except Bobcats (Table 2). Coyotes were active during 24-h periods, with the fewest number of detections between 1400–1500 (n = 8) and the greatest number of detections between 1800–1900 (n = 55). While Bobcats were also active throughout the entire 24-h period, more events were observed during night (n = 77) than day (n = 52). Far more predators were detected on cameras in the playa than in the hills or canyon ($X^2 = 377.1$, df = 2, $P < 0.001$), comprising primarily Coyotes, although notable exceptions were Domestic Dogs and Bobcats found mostly in the hills and common wading birds (Great Blue Herons and Great Egrets) seen in both canyon and playa. Northern marshes also tended to have more predators on camera than southern or Shoshone ($X^2 = 240.9$, df = 2, $P = < 0.001$), with coyotes most commonly detected in northern marshes, but Domestic Dogs were more common in southern marshes and Shoshone, and Bobcats were far more common in southern marshes. Total predator detections varied significantly among seasons (Fig. 2) and was greatest in fall and lowest in winter and spring ($X^2 = 354.4$, df = 3, $P < 0.001$). Coyotes were the most abundant predators in all seasons except spring and were most abundant in fall. Domestic Dogs were absent during summer presumably due to changes

TABLE 1. Number of detections of predator species grouped by guild, as described in text, near Amargosa Vole (*Microtus californicus scirpensis*) habitat in Shoshone and Tecopa, Inyo County, California, between 2013 and 2014. Methods of observation include point count, camera-trap, and incidental observation.

Guild	Common name	Scientific name	Number detected on:		
			Point count	Camera trap	Incidental observation
Aerial					
	American Kestrel	*Falco sparverius*	9		
	Barn Owl	*Tyto alba*	1	4	
	Cooper's Hawk	*Accipiter cooperii*	1	7	
	Common Raven	*Corvus corax*	246	8	
	Great-horned Owl	*Bubo virginianus*	10	1	
	Long-eared Owl	*Asio otus*			3
	Loggerhead Shrike	*Lanius ludovicianus*	6	2	
	Merlin	*Falco columarius*	2		
	Northern Harrier	*Circus cyaneus*	57	4	
	Osprey	*Pandion haliaetus*	1		
	Prairie Falcon	*Falco mexicanus*	7		
	Red-shouldered Hawk	*Buteo lineatus*	3		
	Red-tailed Hawk	*Buteo jamaicensis*	17	3	
	Turkey Vulture	*Cathartes aura*	4		
Waders					
	American Bittern	*Botaurus lentiginosus*	17	5	
	Black-crowned Night-heron	*Nycticorax nycticorax*	1		
	Great Blue Heron	*Ardea herodias*	56	36	
	Great Egret	*Ardea alba*	49	3	
	Green Heron	*Butorides virescens*	1		
	Snowy Egret	*Egretta thula*	5		
Pursuit					
	Coyote	*Canis latrans*	114	593	
	Greater Roadrunner	*Geococcyx californianus*	2	7	3
	Bobcat	*Lynx rufus*		129	
	Kingsnake	*Lampropeltis getula*			4
	Gopher Snake	*Pituophis catenifer*	1		3
	Coachwhip	*Masticophis flagellum*			1
	Spotted Skunk	*Spilogale gracilis*			1
	Desert kit fox	*Vulpes macrotis*			At least 1 (scat)
Invasive					
	American Bullfrog	*Lithobates catesbeianus*	40	2	
	Domestic Dog	*Canis lupus familiaris*	63	27	
	Domestic Cat	*Felis catus*	4		6
Total			714	831	22

in the behavior of owners, and there was little seasonal change in detections of Bobcats.

Point counts yielded 717 predator observations, with predators documented in 40 marsh patches (Table 1). Coyotes and Common Ravens were present in the greatest number of marshes (n = 18) and were the most frequently detected species overall. Over the course of all sample periods, S per marsh ranged from zero to 29 (Table 3). Average S in northern marshes (8.83 ± 9.4 SD) was higher although not significantly so than in southern marshes (5.0 ± 5.0; t = 1.36, df = 35, P = 0.182, Fig. 3); similarly S across regions was not significantly different ($F_{2,36}$ = 1.006, P = 0.376), with mean S in playa of 7.9 ± 8.5, canyon of 14.5 ± 2.1, and hills of 5.0 ± 7.2 (Fig. 3).
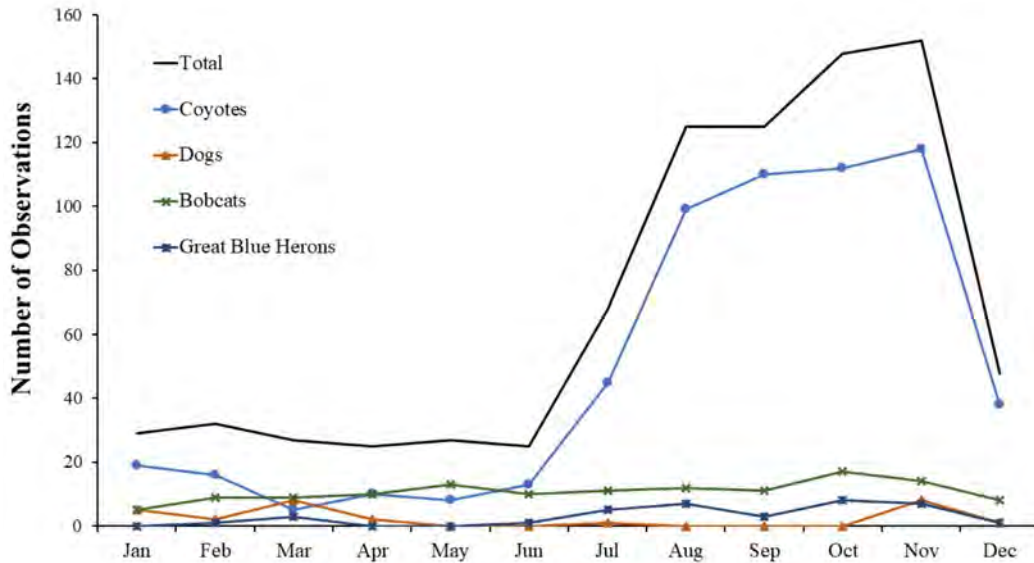
Figure 2. Seasonal patterns (number of independent events captured on camera) of predators of Amargosa Vole (*Microtus californicus scirpensis*) near Tecopa, Inyo County, California, between 2013 and 2014.

Callback surveys yielded 11 observations of Great-horned Owls including six in northern marshes, three in southern marshes, and two in Shoshone. We found 509 predator scat samples in the field, including 285 from Coyotes, 115 from Domestic Dogs, 67 from Bobcats, two from Spotted Skunks, and one from a Desert Kit Fox. We could not identify the rest. We found mammal bones in 219 samples. These scats originated from Coyotes, Domestic Dogs, Domestic Cats, and Bobcats. Thirty-nine owl pellets originated from Long-eared Owls (93.1% of pellets) and Great-horned Owls (6.9% of pellets). Pellets and feces were most abundant in southern marshes (n = 326, 59% of samples recovered), followed by northern marshes (n = 183, 33.4%), and Shoshone (n = 39, 7.1%). There were more scat samples recovered in January (n = 103) and February (n = 101) than all other months, during which we found no more than 53 samples. There were vole remains in one pellet sample, from a Great-horned Owl

(3.6%; n = 29), and 18 scat samples from Coyotes (2.9%, n = 285) and Bobcats (17.5%, n = 67). The prevalence in Bobcats was significantly higher than for Coyotes and Great-horned Owls (X^2 = 16.97, df = 2, P = 0.002).

Discussion

Our survey of predators in the Amargosa River basin reveals a high diversity of 31 species of potential predators of Amargosa Voles, and vole bones in feces from Bobcats and Coyotes and pellets of Great-horned Owls confirm these species as predators. Extensive presence of some wading bird species in bulrush habitat occupied by voles strongly supports their potential as predators of the vole. We show increased predator pressure in autumn, and specific predators occupying differing spatial patches.

Among confirmed vole predators, Bobcats had high prevalence of vole bones within scat but were found only

Table 2. Differences in predator detections, inferred from independent observations on camera-traps near Amargosa Vole (*Microtus californicus scirpensis*) habitat in Shoshone and Tecopa, Inyo County, California, between 2013 and 2014. The most common wading bird species were Great Blue Herons (*Ardea herodias*) and Great Egrets (*Ardea alba*). Descriptions of how day and night, northern and southern, marsh region, and season are differentiated are provided in the text.

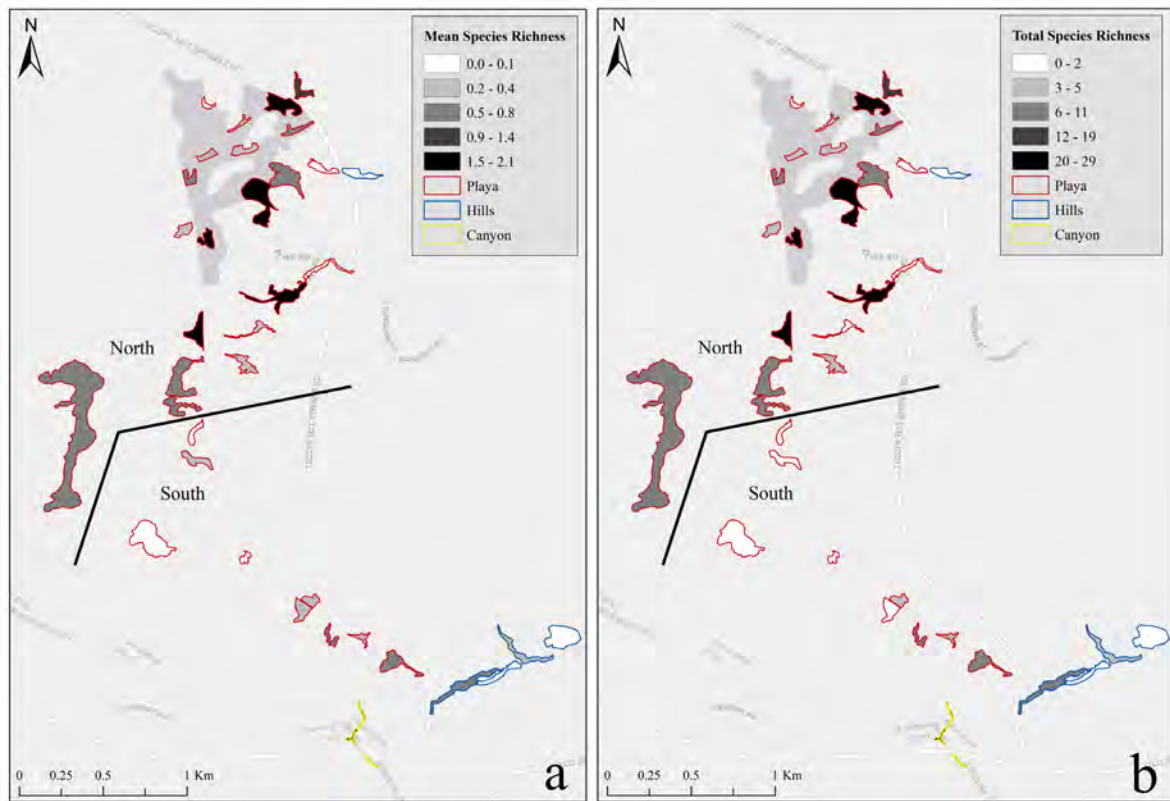|	All species combined	Common Raven	Coyotes	Domestic Dogs	Bobcats	Northern Harriers	Common wading birds
Day	472	8	320	27	52	4	39
Night	359	0	273	0	77	0	0
Canyon	46	0	14	0	1	0	22
Hills	282	2	125	22	128	0	0
Playa	503	6	454	5	0	4	17
North marshes	464	2	424	4	0	4	17
South marshes	268	6	118	9	95	0	22
Shoshone	99	0	51	14	34	0	0

Figure 3. Yearly mean (left panel) and total (right panel) species richness of data collected monthly in marshes classified by region (North or South) and habitat type (Playa, Hills, Canyon) sampled through point count methods. Sampling occurred near Tecopa, Inyo County, California, between 2013 and 2014.

in a few marshes in the hill region. We only detected this reclusive species using baited cameras, and such cameras were only used in southern sites where habitat heterogeneity is greater than in the north. We cannot differentiate a detection bias because of baited cameras or whether Bobcats do not occur in northern marshes we did not bait. The most abundant non-native predator was Domestic Dogs with incidental Domestic Cats, highlighting potential anthropogenic threats to the Amargosa Vole.

Geographical patterns ranged from Shoshone, where habitat is being restored for future vole introduction, with 16 species of predators, including two snake species found only in Shoshone, to Tecopa where predator species richness tended to be higher to the north. Northern marshes tend to be more numerous, larger, and possibly more productive which could allow for coexistence of some species (Brown 1981). Total predator detections were also more numerous in the north. Voles are unlikely to move among northern and southern marshes based on genetic data (Krohn et al. 2017) although most predators we recorded would be able to readily move between northern and southern areas.

Habitat associations reflected different predator preferences as well as risk of predation for voles. Thick bulrush litter as well as tall, dense cover of live bulrush are very important cover for Voles (Klinger et al. 2015). Nevertheless, pursuit predators were abundant, and

bulrush litter likely does not completely protect voles from being captured by Coyotes and Bobcats. Predator species richness was higher in the Amargosa River canyon than playa or hills, while total detections were higher in playa marshes than hills or canyon. Predator species richness was likely influenced by habitat heterogeneity, and in this

Table 3. Summary statistics associated with predator observations using point counts (direct observation) near Amargosa Vole (*Microtus californicus scirpensis*) habitat in Shoshone and Tecopa, Inyo County, California, between 2013 and 2014. Species richness is indicated by S: PSA = Predator S for all species, AS = Aerial S, WS = Wading bird S, PS = Pursuit S, IS = Invasive S, and PO = number of point count predator observations of predators. Predator guild, marsh grouping by north and south, region, and presence or absence or voles and bulrush are explained in text.

	Guild of Predator					
Sites	PSA	AS	WS	PS	IS	PO
Northern	19	10	6	1	3	413
Southern	16	9	2	2	3	236
Shoshone	9	5	0	2	2	66
Playa region	20	11	5	1	3	478
Hill region	8	5	0	1	2	174
Canyon region	8	3	1	2	2	63
All marshes	25	13	6	2	3	717

regard both southern sites and those in the canyon have the greatest diversity due to the close juxtaposition of wetland, upland, and edge plant communities. Domestic Dogs were uncommon on playa, which we expected as most homes in the area are in the hills, relatively distant from the playa. Possible spatial differences could also be influenced by intraguild interactions as described previously (Fedriana et al. 2000).

Our study examined daily and seasonal trends in predator detections. Aerial and wader predators were more abundant during the day. Nocturnal pressure from raptors was likely underestimated because of our use of a limited number of calls of owl species, reluctance of some species to call back, and our inability to see most raptors at night. Coyotes were common day and night. The majority of the predators are resident, while migrants such as Long-eared Owls, Osprey (*Pandion haliaetus*), and Black-crowned Night-heron (*Nycticorax nycticorax*), and seasonally active predators such as Gopher Snakes and Coachwhips, were not observed during much of the year. Overall seasonality of predator detections favored the fall (comprising predominantly Coyotes and herons), during the period when vole populations are in the downward phase of their population cycle (Foley and Foley 2016, Janet Foley et al., unpubl. report). It is unlikely that this is a delayed numerical response (given the annual birth pulses and long generation times of most predators) but rather a behavioral response. Predators may be recruited to the area throughout the time of abundant prey due to high birth rates of small mammals in the summer but then remain and even increase into the fall even as prey resources diminish. In addition, predator activity may appear increased if they are more visible because they are spending more time hunting in the face of reduced food.

The question of whether these predators are regulating voles or possibly overharvest voles and diminish population viability is very important for vole conservation. All of the seven most ubiquitous species (Ravens, Coyotes, Domestic Dogs, Great Egrets, Great Blue Herons, Northern Harriers, and Bobcats) are generalists, and we did not detect specialist predators (e.g., weasels, *Mustela* sp.). Among studies examining how microtine populations are influenced by top-down regulatory factors (Korpimaki et al. 2002; Banks et al. 2004), many are done at high latitude and examine voles subject to specialist predators. Instead, the Amargosa Vole occurs at one of the lowest latitudes in which predation on microtines has been studied. Further work would be helpful to clarify specific vole-predator interactions, as incidence, abundance, and proportional frequency of a species are not necessarily indicators of potential impacts. For instance, there was an approximately 700% increase in Common Ravens in the western Mojave Desert from 1969 to 2004 (Boarman and Kristan 2006), with a shift toward individuals becoming permanent residents due to human-provided

food, water, and nest sites (Knight et al. 1993; Boarman and Berry 1995). Despite this, and their relatively high frequency in our sampling, we have no evidence that they are a meaningful predator on Amargosa Voles. In contrast, American Bitterns (*Botaurus lentiginosus*) were of moderate frequency in our samples, but this is likely because they are cryptic and less likely to be detected by of our survey methods. Nevertheless, bitterns have disproportionally greater effects on voles than our data would indicate, as we have directly observed bitterns hunting and even capturing voles.

Our study is preliminary and narrow in temporal scope, and our assessment of predator activity is unfortunately not paired with detailed data from the same space and time on vole numbers. The fact that baited cameras were used in the only two marshes where we were confident there were no voles was a bias in that more predators were likely observed because of the bait, precluding us from comparing numbers between marshes with and without bait and (or voles). Some methods, such as scat surveys, which were conducted monthly, may have underestimated predators, although in this highly arid environment, we have observed scat persisting for multiple months. Nevertheless, there is a very large number of potential predators on the Amargosa Vole, some of which may have a strong influence on vole population dynamics and demography, particularly Coyote, Bobcat, American Bittern, Great-blue Heron, Great Egret, Northern Harrier, and owls. Our study provides valuable baseline data for assessing potential top-down influences on Amargosa Voles, in support of earlier writers emphasizing the need for such research to manage this species (USFWS 1997; Leroy McClenaghan and Stephen Montgomery, unpubl. report).

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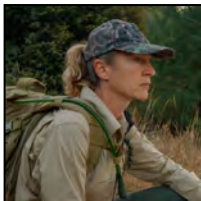
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Wrestling for Real Estate: Male-male Interactions in Breeding Foothill Yellow-Legged Frogs (*Rana boylii*)

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Abstract.—Male-male aggression is common among anurans competing for females at traditional breeding sites. We describe two male-male behavioral interactions observed between male Foothill Yellow-legged Frogs (*Rana boylii*) at a surrogate breeding habitat, with two different outcomes.

Key Words.—aggression; anurans; amplexus; behavior; breeding; competing, dam, lek, surrogate

For many anurans from temperate climates, annual congregations at breeding sites are a cauldron of male-male competition for mates (Wells 1977, 2007). The arrival of females intensifies selective pressure on the more numerous males to find a mate quickly, often pressuring males to grasp anything in an amplexal hold, including a different species, and sometimes even inanimate objects (Wells 2007; sensu Alvarez 2011). Males often mistakenly grasp conspecific males, but in most species the amplexed male emits a release call that is recognized by the grasping male, which often results in a timely release from amplexus (Wells 1977; Simović et al. 2014). Males may also aggressively defend display sites, often calling from stationary positions, which may include territories, or resources such as oviposition sites (Howard 1978; Wells 2007).

Foothill Yellow-legged Frogs (*Rana boylii*) historically inhabit coastal streams and rivers from Marion County, Oregon, to Ventura County, California, and inland along the Sierra foothills, south to Kern County, California (Zweifel 1955). They have a lek mating system and often return to certain lek sites within a stream year after year (Kupferburg 1996; Wheeler 2007). Males establish and defend territories within oviposition sites, calling from underwater (Davidson 1995; MacTague and Northen 1993) or from emergent rocks, where they shift position to vary the direction of their acoustic message. Male-male aggression in this species includes males vocally and physically defending specific sites that include above-water calling substrates within a breeding area (Stebbins 1951; Zweifel 1955; Wheeler 2007; Wheeler and Welsh 2008; Silver 2018). Wheeler and Welsh (2008) observed that calling by males elicited aggression from other males, and that males with a larger territory size engaged in more aggression interactions than males with smaller territories. Observed aggression included acoustic dueling, wresting, chasing, or a combination thereof (Wheeler and Welsh 2008). Here, we describe observations of two types of male-male interaction in the unusual setting (atypical breeding habitat) of a small concrete dam impounding an artificial reservoir, a habitat not considered typical for a stream obligate such as *R. boylii* (Jeff Alvarez and Jeffery Wilcox, unpubl. data).

Stewart Pond is a privately owned reservoir in eastern Sonoma County, California, used for irrigating vineyards and for recreation. At full capacity, the reservoir surface area is approximately 0.8 ha and it reaches a maximum depth of 8 m. A dam (approximately 10 m long and 1 m tall) on the northwest side of the reservoir impounds the water. The dam has a recessed, extended block that directs the water over the center of the dam (Fig. 1). On the evening 20 April 2017, while conducting a control project for American Bullfrogs (*Lithobates catesbeianus*; Jeff Alvarez et al., in review), we stopped at the outfall side of the dam to remove three *L. catesbeianus* from the outflowing stream at the base of the dam. After removing the bullfrogs, we withdrew to the nearby bank to process them and await nightfall. At the base of the dam, we noticed five *R. boylii*, three on the concrete footing and two at the top of the dam. The frogs (all approximately 40–50 mm snout-urostyle length) were on either side of the flow over the dam: some above, and some below (Fig. 2). Suddenly, one of the *R. boylii* turned to face the vertical dam wall, leapt approximately 0.3 m up the concrete face and began climbing. The dam face had an undulating surface from corrugated metal roofing material used to form the concrete during construction, and the wet surface was covered in a film of algae. Nevertheless, the frog was able to slowly climb the slick surface toward the top of the dam (Fig. 2). Near the top of the dam it slipped and fell, but repeated the effort and made it to a ledge in the face of the dam. The frog sat on the ledge for less than 1 min, and then suddenly leapt approximately 0.6 m to the top of the dam, landing on top of another, slightly larger, *R. boylii* that had been vocalizing.

The leaping frog (which had no line of sight on the vocalizing frog prior to jumping) immediately grasped the vocalizing frog in a partial amplexal hold (Fig. 3) such that it positioned its left front leg in pectoral amplexus and its right front leg in cephalic amplexus of the bottom frog, facing the posterior end of the bottom frog. What

Figure 1. Stewart Pond, a reservoir created by impounding three intermittent streams, at the foot (western slope) of Mount Saint Helena in Sonoma County, California. The dam measures approximately 1 m tall and 10 m wide. (Photographed by Jeffery T. Wilcox).

ensued was a 20-min wrestling match, with the bottom frog trying to throw off the top frog. Sometimes the amplexed (lower) frog vibrated violently, interspersing the vibrations with low croaking, and other times it spun, bucked, and tried to wipe the frog off with its right rear leg (Fig. 3). Bursts of struggling activity were followed by periods of inactivity, which presumably afforded a rest period. Over the 20 min, the scuffling frogs covered an area of approximately 0.5 m. They moved from the recessed area of the dam, nearly tumbled over the dam face, and then moved to the top of the dam and out of the water (Fig. 4).

Concurrent with the activities described above, on the other side of the dam another *R. boylii* climbed the dam face (Fig. 4) and amplexed a non-calling conspecific. In this case, the second frog was already situated on the ledge, so the climbing frog was able to see it before pouncing on it from a much shorter distance. In this interaction, no wrestling ensued, but the amplexed frog did vocalize and vibrate. After about 2 min, the amplexing (top) frog dismounted and moved a short distance away. After 20 min of observations, we left to continue catching *L. catesbeianus*. We returned after approximately 1 h had passed to find only one *R. boylii* on the dam. Therefore we do not know the fate of the first pair that we observed.

Our observations suggest that male *R. boylii* may use concrete dam sites as they woud traditional lek sites in unaltered streams. In the first encounter, the length of the wrestling match suggests that the aggression was perhaps for the calling site itself. Both males vibrated and vocalized during the struggle, yet neither yielded during the observation. In the second, short-duration interaction, only the amplexed (bottom) male vocalized and vibrated, suggesting an incident of misdirected amplexus; the release call signaling the amplexing frog to release its hold.

Rana boylii populations are in decline in many areas of their native range due to the construction of
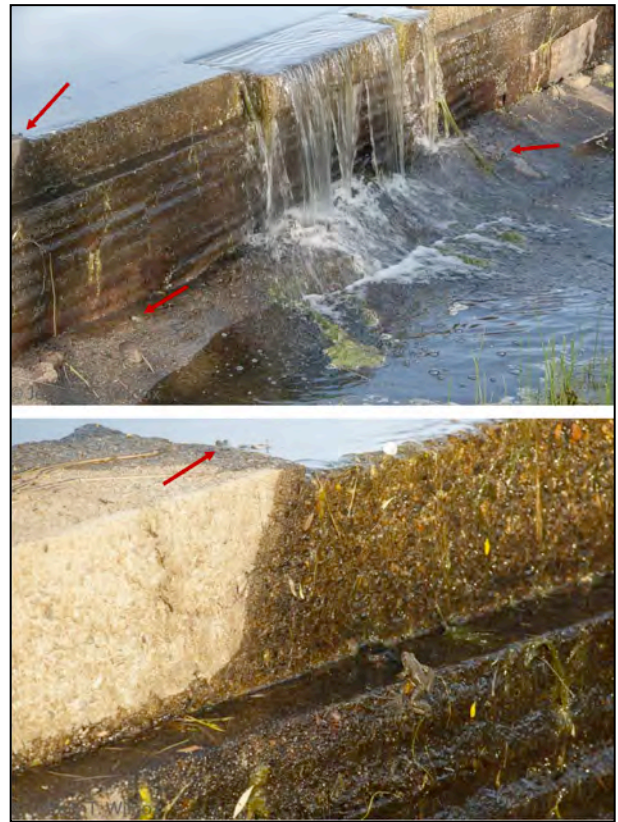
Figure 2. (Top) Locations (arrows) of three of the four Foothill Yellow-legged Frogs (*Rana boylii*) that were observed engaging in combative behaviors on the dam of Stewart Pond, Sonoma County, California. The upper left frog was vocalizing at the time of the photograph and prior to being amplexed by a second male. (Bottom) One of two *R. boylii* that climbed the vertical face of the concrete dam on Stewart Pond Sonoma County, California. Upon reaching the shelf, it settled for a moment and then leapt over the dam and on to the back of the calling male (arrow), apparently locating this conspecific by sound. (Photographed by Jeffery T. Wilcox).

large reservoirs and the regulation of flows from those impoundments (Lind 1996, 2005; Kupferburg et al. 2012). Recent reports, however, reveal that *R. boylii* may exhibit behavioral plasticity in using small reservoirs for mating, oviposition, tadpole rearing, and as dispersal habitat once non-native predators and competitors are removed (Jeff Alvarez and Jeffery Wilcox, unpubl. data). In the 3 y after numbers of *L. catesbieanus* were greatly reduced at Stewart Pond, we observed *R. boylii* egg masses, two generations of tadpoles, and three cohorts of post-metamorphic *R. boylii* (Jeff Alvarez and Jeffery Wilcox, unpubl. data). Our observations indicate that small concrete dams may be suitable as surrogate calling substrates for calling male *R. boylii*, and that *R. boylii* are excellent climbers; able to climb wet, slick, vertical surfaces. The climbing ability we documented may have management implications in situations where anthrogenic structures, such as dams and wiers, are placed within habitat used by *R. boylii*. The ability of this species to

FIGURE 3. (Top) Position of the amplexing male-male Foothill Yellow-legged Frogs (*Rana boylii*), which was followed by apparent combative behavior. Shortly after landing on the bottom male, the top male *R. boylii* appeared to grasp the bottom male, both frogs vocalizing during the struggle. Note the inflated vocal sacs of the top frog; concentric rings in the water indicate the pulsations of the vocal sacs of the lower frog. (Bottom) Combative behavior between two male *R. boylii*. Note the rippling water, which is produced by vibrations of the amplexed (lower) conspecific, which also used vocalization and rear leg movements to dislodge the amplexing (upper) male. (Photographed by Jeffery T.Wilcox).

climb should be considered prior to placing structures in riparian areas potentially occupied by this frog.

Acknowledgments.—We are grateful to the Peter Michael Winery for access to Stewart Pond in support of this and other projects.

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FIGURE 4. (Top) Two male Foothill Yellow-legged Frogs (*Rana boylii*) in combat to the extent that they nearly fell over the face of the dam on Stewart Pond. (Bottom) A third male *R. boylii* (extended hind leg), climbed the dam face to pursue and combat a fourth male *R. boylii* (center right). (Photographed by Jeffery T. Wilcox).

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Growth Curve of White-tailed Antelope Squirrels From Idaho

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Abstract.—Daytime rodent trapping in the Owyhee Desert of Idaho produced a single diurnal species: the White-tailed Antelope Squirrel (*Ammospermophilus leucurus*). I found four females that were pregnant and took them back to my laboratory to give birth and I raised their litters in captivity. Litter size ranged from 10 to 12 pups. The pups were born weighing 3–4 g, with purple skin color and with the eyes closed. Pups were successfully weaned at 60 d of age and approached the adult body mass of 124 g at 4 mo of age.

Key Words.—*Ammospermophilus leucurus*; Great Basin Desert; growth; Idaho; Owyhee County

The White-tailed Antelope Squirrel (*Ammospermophilus* leucurus; Fig. 1) is indigenous to a large segment of western North America, from as far north as southern Idaho and Oregon (43° N) to as far south as the tip of the Baja California peninsula (23° N; Belk and Smith 1991; Koprowski et al. 2016). I surveyed a small part of the northernmost extension of the range of the species in Owyhee County, Idaho, south of the Snake River, which initially I searched for likely habitat. Positive confirmation was obtained of a considerable amount of undisturbed desert habitat and I sighted squirrels at and near localities previously recorded in museum databases (vertnet.org). The area surveyed was about 200 km² in the Owyhee Desert, south of Oreana (43°00'N, 116°20'W, elevation 850 m). Much of the Owyhee Desert is public land managed by the U.S. Bureau of Land Management. The soils and vegetation were typical of arid lowland scrub within the Great Basin Desert, with sparsely distributed shrubs, predominantly Great Basin Sagebrush (*Artemisia tridentata*) and Greasewood (*Sarcobatus vermiculatus*). The ground squirrels were strongly associated with the more alkaline soils dominated by Greasewood.

Once I confirmed the existence of suitable habitat, I set out Sherman live traps (model LFAGTD, H.B. Sherman Traps Inc., Tallahassee, Florida) in mid-December and again in early April. I captured one female and two males 12–13 December 2016 and four females and five males 1–2 April 2017. I took the animals into the laboratory immediately upon capture. White-tailed Antelope Squirrels were the only vertebrate animals I captured in the traps. I prevented unintentional capture of nocturnal rodents by closing the traps at night. Although other diurnal rodents, such as Least Chipmunks (*Tamias minimus*), Townsend's Ground Squirrels (*Urocitellus townsendii*), and Belding's Ground Squirrel (*Urocitellus beldingi*) are reportedly present in southwestern Idaho (Groves et al. 1997), I did not find them in this transect.

The average body mass of 12 White-tailed Antelope Squirrels I captured in December and April was 108 g (91–122 g). The four females I captured in the spring were pregnant but were early in their pregnancies, and the body masses of the seven males and five females did not differ significantly ($t = 0.715$, df = 10, $P = 0.503$). After four months in the laboratory (after parturition and lactation for the four pregnant females), average body mass stabilized at 124 g (106–145 g).

The four pregnant females were left undisturbed in individual polypropylene cages with wire tops (36 cm length, 24 cm width, 19 cm height). The cages were lined with cellulose bedding (Comfort Bedding, BioFresh, Ferndale, Washington) and were kept inside light-tight, ventilated chambers maintained under a light-dark cycle with 12 h of light per day (12L:12D) in a room kept at 25° C with 25% relative humidity. I provided nest-building material of thin strips of crinkled paper (Crink-l'Nest, Andersons Lab Bedding Products, Maumee, Ohio) and square cotton pads (Signature Care, Better Living Brands, Pleasanton, California). I provided Purina rodent chow (Rodent Diet 5001, Lab Diet, St. Louis, Missouri) *ad libitum* on the metal cage top, which also held a water bottle with a sipping tube extending into the cage. I provided fresh carrot slices and grapes three times a week.

The four females gave birth 14–19 April. The litter sizes were 10, 10, 10, and 12, which fall within the range of six to 14 reported for a sample of 31 litters from Oregon (adjacent to Idaho) as assessed by autopsy (Whorley and Kenagy 2007). The pups were born with purple skin

Figure 1. White-tailed Antelope Squirrel (*Ammospermophilus leucurus*) from Owyhee County, Idaho. (Photographed by Roberto Refinetti).

FIGURE 2. Development of White-tailed Antelope Squirrel (*Ammospermophilus leucurus*) pups in captivity: (Left) Postnatal day 2, (Middle) day 24, (Right) day 36. (Photographed by Roberto Refinetti).

color and with the eyes closed (Fig. 2). By 9 d of age, the skin began to show some darker pigmentation. By day 13, the eyelids became conspicuous but remained closed. By 19 d, a light, soft body pelage was noticeable. By 24 d, body stripes were showing along the flanks (Fig. 2). After 30 d, the eyes were opened, and pups began to venture out of the nest. After 35 d, the pups were exploring the cage more extensively and starting to nibble on solid food (Fig. 2). After 40 d, all pups moved extensively around the cage and consumed solid food. All pups were successfully separated from their mothers at postnatal day 60, when body mass ranged from 66 to 95 g. I provided grapes *ad libitum* to the recently weaned animals to ensure appropriate hydration.

I initially weighed pups in groups and later individually in intervals of 11 d. The growth curve based on weight gain for squirrels from Oregon is similar to the curve by Pengelley (1966) from four litters raised in the laboratory by females trapped in the Mojave Desert in California (Fig. 3). Both curves show an acceleration of growth around day 40, when the pups start to eat solid food. The similarity of the two curves reveals little difference in the growth of captive White-tailed Antelope

Squirrels from the Mojave and Great Basin deserts. As is the case in primates (Altmann et al. 1981; Cheverud et al. 1992; Bolter and Zihlman 2003), it is likely that captive individuals have accelerated rates of growth in comparison to their wild counterparts.

Litter size cannot be reliably compared because of the small number of litters in both cases, but the litter size reported by Pengelley (1966) in California (range, 8–11 pups) was slightly smaller than what I found in Idaho (range, 10–12 pups). In the wild, mean litter size is known to be smaller in California (7.4) than in Oregon (9.3) and even smaller further south in Baja California (5.9; Whorley and Kenagy 2007). Benchmarks of development, such as the timing of the onset of skin pigmentation and opening of the eyes, were also similar to observations for California squirrels (Pengelley 1966), except that the Idaho animals opened their eyes, and later could be weaned, about 5 d ahead of the California animals. In general, my observations from Idaho are in agreement with observations from California that the young of White-tailed Antelope Squirrels are born in larger litters and develop more slowly than the young of three other ground squirrel species, Golden-mantled Ground Squirrel (*Callospermophilus lateralis*), Mohave Ground Squirrel (*Xerospermophilus mohavensis*), and Round-tailed Ground Squirrel (*Xerospermophilus tereticaudus*; Pengelley 1966).
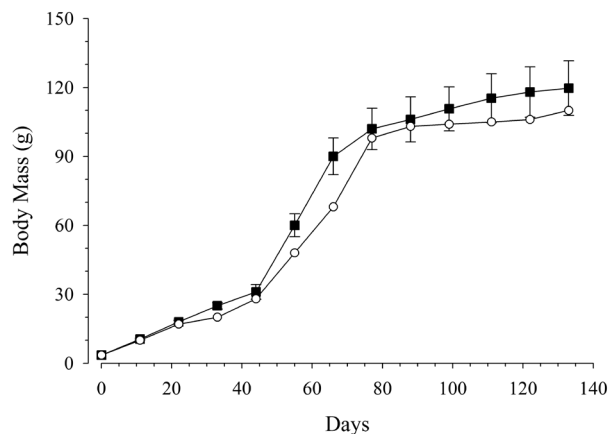
FIGURE 3. Growth curve of White-tailed Antelope Squirrels (*Ammospermophilus leucurus*) in captivity. Closed squares: data from this study (means ± SD). Open circles: data from Pengelley (1966).

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Notes

Recent Observation of Kit Fox (*Vulpes macrotis*) Outside of Its Known Range in Oregon

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Abstract.—Here we report on an observation of a Kit Fox (*Vulpes macrotis*) outside of its known range, near Ontario, Oregon.

Key Words.—distribution; Great Basin Desert; roadkill

On 15 February 2017, we collected an apparent vehicle-caused mortality of a male Kit Fox (*Vulpes macrotis*) on Interstate Highway 84 in eastern Oregon, USA (Fig. 1). The specimen was located on the highway near the crossing of the Snake River and the border with Idaho (UTM zone 11T 0504666, 4872846; datum: NAD 83). We used a 3-mm biopsy punch to collect an ear-tissue sample from the carcass to confirm species identification as Kit Fox using DNA (Benjamin Sacks, pers. comm.). We also extracted a tooth for determining age, and based on cementum annuli analysis (Matson's Laboratory, Manhattan, Montana), this individual was 2 y old.

Historically, Kit Foxes ranged from southeastern Oregon and southwestern Idaho south through the deserts and semi-arid regions of California, Nevada, Arizona, and New Mexico, into northcentral Mexico (McGrew 1979). The current distribution is somewhat unknown, although in Oregon, recent research (Milburn and Hiller 2013; Vesely, D. 2015. Conservation assessment of the Kit Fox in Southeast Oregon. Oregon Wildlife Institute, Corvallis, Oregon, USA. https://www.fs.fed.us/r6/sfpnw/issssp/documents3/ca-ma-kit-fox-in-oregon-2015-08-508.pdf. [Accessed 20 April 2018]; Eckrich et al. 2018) has confirmed the species presence in southeastern Harney and southwestern Malheur counties, west and southwest of Burns Junction in southeastern Oregon (Fig. 2). Prior to those studies, the most recent confirmed detections occurred in 1993 (Keister 1994). An unusual past record indicated an individual present in Klamath County (Laughlin and Cooper 1973). Recent (1992–2012) observations in Idaho are generally restricted to the southwestern portion of the state (Idaho Department of Fish and Game. 2016. Idaho Fish and Wildlife Information System, Species Diversity Database, Idaho Natural Heritage Data. https://idfg.idaho.gov/species/taxa/17479. [Accessed 3 May 2018]; Fig. 2). Recent harvest records from Nevada confirm presence in all northern-tier counties (Russell Woolstenhulme, unpubl. data). Our recent observation was located about 175 km from the known recent (2012–2015) detections in Oregon, and about 110 km from nearest recent detection in Idaho. We cannot determine whether the individual kit fox described here was a wild or a captive individual, a dispersing individual, or a potential indication of range expansion. We recommend additional efforts, however, to study the distribution and ecology of this species in Oregon (and elsewhere) because the Kit Fox is a species of conservation concern in Oregon (e.g., Vesely 2015. *op. cit.*; http://www.oregonconservationstrategy.org/).

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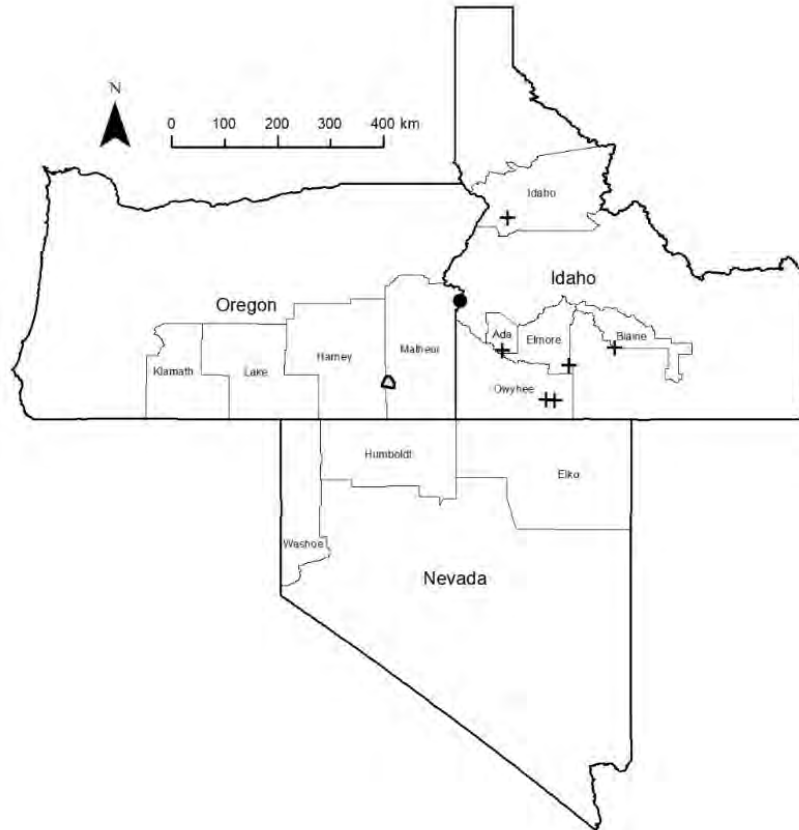
Figure 2. Recent detections of Kit Foxes (*Vulpes macrotis*) in Idaho (1992–2012), Nevada (northern-tier counties based on harvest data), and Oregon (gray polygon = 2012–2015, closed circle = 2017), USA.

DIET OF THE SAN JOAQUIN ANTELOPE SQUIRREL IN THE SOUTHERN PORTION OF ITS RANGE

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Abstract.—During studies of distribution and population density of the San Joaquin Antelope Squirrel (*Ammospermophilus nelsoni*), I collected fecal samples from two sites in the core of the current distribution of the species: The Paine Preserve of The Nature Conservancy on the floor of the San Joaquin Valley, and the Elkhorn Plain, an upland site on the southwestern edge of the range of the species. A commercial analysis of these samples showed that the San Joaquin Antelope Squirrel was omnivorous, consuming a wide variety of foods including leaves, fungi, seeds, arthropods and occasionally other animal matter. The breadth of food types here was greater than that previously documented, though not unusual for a sciurid. Plant material consumed included the foliage, stems and seeds of grasses, herbs, and shrubs. Non-native plants, such as the grasses *Bromus* sp. and Arabian Grass (*Schismus arabicus*), and the herb Storksbill (*Erodium cicutarium*), were a significant proportion of the diet.

Key Words.—*Ammospermophilus nelsoni*; forbs; grasses; San Joaquin Desert; shrubs

INTRODUCTION

The San Joaquin Antelope Squirrel (*Ammospermophilus nelsoni*) is a 140-180 g diurnal rodent (Fig. 1) found in the San Joaquin Desert (Germano et al. 2011) of the southern San Joaquin Valley of California (Best et al. 1990; Williams 1981). The species is listed as Threatened by the California Fish and Game Commission (California Department of Fish and Game 1980). Its elevational distribution spans from about 50 to 1,100 m and it may be found on open terrain and gently sloping hills (Best et al. 1990). Habitats occupied are open perennial bunchgrass, non-native annual grasslands, Saltbush (*Atriplex*) scrub, Jointfir (*Ephedra*) scrub, and other open shrublands on sandy loam and alluvial soils. Cultivated croplands are avoided (Grinnell and Dixon 1918; Hawbecker 1953; Williams 1981). Although shrubs may provide significant thermal cover (Hawbecker 1953; Heller and Henderson 1976), San Joaquin Antelope Squirrels are also found in areas devoid of shrubs (Best et al. 1990; Harris and Stearns 1991).

Little is known concerning the diet of this species and the relation of diet to habitat use. The only prior study of diet is that of Hawbecker (1947). That study was conducted in the Panoche Hills, San Benito County, California, near the northern range limit of the species. The study area was dominated by non-native annual grassland, with few to no shrubs. During studies of distribution and population density (Harris and Stearns 1991), I collected fecal samples from two sites in the southern portion of the geographic range of the species. In this article, I describe the results of fecal analysis from the two study areas and relate the results to the characteristics of the occupied habitats. The two sites are compared to each other and to the results of Hawbecker (1947).

METHODS

Study sites.—I collected fecal samples at two locations: a 13-ha live-trapping study plot located on the Elkhorn Plain, San Luis Obispo County, and at the Paine Preserve in Kern County. The Elkhorn Plain is situated between the Carrizo Plain and the Temblor Range. The topography of the study area is rolling terrain gently sloping upward to the east. Vegetation at the site was dominated by grasses and widely spaced shrubs, primarily California Jointfir (*Ephedra californica*), with a few Anderson's Boxthorn (*Lycium andersonii*). Smaller Yellow Aster (*Eastwoodia elegans*) and buckwheat (*Eriogonum* spp.) were common. Herbaceous cover was dominated by the introduced annual grass Arabian Grass (*Schismus arabicus*) although a number of other grasses and herbs were present.

FIGURE 1. San Joaquin Antelope Squirrel (*Ammospermophilus nelsoni*) from the Panoche Hills, San Benito County, California. (Photographed by John H. Harris).

The second location was a 5.6-ha live-trapping study plot located at the Paine Preserve of The Nature Conservancy. This site, on the floor of the San Joaquin Valley, was a low-lying area dominated by Spinescale Saltbush (*Atriplex spinifera*) and Shrubby Seablight (*Sueda fruticosa*). Low hummocks were separated by seasonally flooded alkali flats. Most squirrel captures and all burrowing activity were located on these low mounds or on road banks. I collected fecal samples at various times from the end of May through mid-July, so any annual vegetation was dry as one would expect (Germano et al. 2011). Rainfall at the Bakersfield Airport weather station was 141 mm (86% of normal) for 1987–1988 and 95 mm (58% of normal) for 1988–1989; these were the two rainfall years preceding sampling (https://www.weather.gov/hnx/bflmain).

Sample collection.—Characterization of the diet based on fecal samples has been shown to be comparable to stomach samples for several species of ground squirrels, including the Columbian Ground Squirrel (*Urocitellus columbianus*; Harestad 1986), Black-tailed Prairie Dog (*Cynomys ludovicianus*; Wydeven and Dahlgren 1982), and Townsend's Ground Squirrel (*Urocitellus townsendii*; Van Horne et al. 1998), making this an appropriate method for a protected species that should not be sacrificed for gastro-intestinal contents. I collected fecal samples at the Elkhorn Plain site at three times: 31 May to 1 June 1988 (24 samples), 7–8 July 1988 (18 samples), and 27–28 May 1989 (50 samples). At the Paine Preserve study site, I collected fecal samples 9–10 June 1988 (13 samples) and 13–14 July 1988 (five samples). I collected fecal samples from Sherman XLK live-traps that were baited with commercially available mixed bird seed and peanut butter. I emptied traps after each capture to avoid mixing of feces from different animals. I stored fecal samples in plastic bags with no residual bait from traps. Later, I transferred the dried samples to labeled paper envelopes and I kept them in dry conditions thereafter.

Sample analysis.—I selected three random fecal pellets from the first capture of each individual to standardize the overall amount of fecal material between samples. Using the three fecal pellets, a slide was prepared by Composition Analysis Laboratory, Inc., Fort Collins, Colorado, in the following manner (Hansen et al. (1973): the feces were crushed or ground and then cleared with a bleach solution. The resulting fragments were washed through a 200 mesh Tyler standard screen (0.074 mm openings), which removes solubles and extremely small, unidentifiable particles. The washed material was then transferred to a microscope slide, making an effort to ensure that roughly equal amounts of material were on each slide.

Plant materials in the fecal samples were identified by experienced technicians using the characteristics of leaf epidermis or seed coats. I provided the Composition

Analysis Lab with a plant list for each site and a list of the species in the bait seed mix. The importance of various items in the diet was quantified in two ways. For each sample slide, 20 microscope fields of view were examined. The occurrence of individual food types was recorded for each field. The percentage frequency among microscope fields was then calculated. This frequency can be converted to density if certain assumptions are made (Hansen et al. 1973; Sparks and Malechek 1968), including equal fragment size and the random distribution of plant fragments over the slide. Once the density of fragments was estimated, the percentage relative density was calculated by dividing the density of fragments of a particular species by the density of fragments of all species. The results provided by the commercial laboratory included the percentage relative density of each food type in each sample. I report the means of these relative densities for each location and sampling date, which provides an estimate of the proportion of the diet made up by a particular food type. I also calculated the percentage frequency of occurrence: the percentage of samples from a given site and time period in which a food type occurred for each food type for each site and time period.

RESULTS

Composite and grass seeds were found in nearly every sample (Table 1). The results for the Paine Preserve site for 9–10 June 1988 indicate that seeds made up the largest proportion (58.8%) of the diet (Table 2). Even at this date, when the site was very dry, leaves made up an appreciable proportion (16.2%) of the diet. The only leaves that were found in a high proportion of the samples were *Atriplex* leaves. Fungal fragments made up a similar proportion (17.2%) of the diet and arthropods composed the remaining 7.5%. The Paine Preserve samples I collected about one month later (13–14 July 1988) revealed a somewhat different pattern. Arthropods made up the largest proportion (41.2%) of the diet and were found in all samples. Seeds had decreased in the proportion (39.1%) of the diet. The category denoted as seed includes unknown seeds, possibly including bait. If this category is eliminated, the proportions of arthropods and green vegetation would be slightly increased. Leaves continued to make up a significant fraction of the diet during this later sampling period (19.9%) but fungi were absent (Table 2).

At the Elkhorn Plain site, seeds were the largest diet category during all three sampling periods, but this category was dominated by the unknown seed category, leaving considerable uncertainty regarding the true importance of seeds at this site. Identified seeds which I found in the diet included grass seeds, chenopod seeds, and seeds of *Lycium andersonii*. Leaves were important during the 31 May to 1 June 1988 sampling period (22.4%). The most important plant foods eaten at this time included the leaves of grasses (*Bromus* sp. and

TABLE 1. Percentage frequency of occurrence of food types in the diet of the San Joaquin Antelope Squirrel (*Ammospermophilus nelsoni*). The percentage of samples in which a given food type was found is given for each study site and season. The abbreviation n is the number of individual squirrels from which fecal samples were collected. The locations of study areas and time of sample collection are given in the text.

	Paine Spring 1988 n = 13	Paine Summer 1988 n = 5	Elkhorn Spring 1988 n = 24	Elkhorn Summer 1988 n = 18	Elkhorn Spring 1989 n = 50
Seeds					
Grass	92.3	60.0	12.5	44.4	32.0
Chenopod	15.4	60.0	4.2	0	0
Composite	84.6	80.0	0	5.6	2.0
Lycium spp.	0	0	50.0	22.2	18.0
Unknown seed	0	40.0	100.0	89.9	100.0
Unknown pod	0	0	0	16.7	0
Leaf Tissue					
Allenrolfea	7.7	0	0	0	0
Atriplex	76.9	40.0	8.3	0	0
Baileya	0	20.0	0	0	0
Eremalche	30.8	0	12.5	0	0
Bromus	0	0	50.0	5.6	4.0
Schismus	0	0	62.5	0	10.0
Distichlis	0	0	0	0	2.0
Oryzopsis	0	0	0	0	2.0
Chrysothamnus	0	0	0	5.6	0
Trifolium	0	0	0	0	2.0
Oenothera	23.1	0	4.2	0	2.0
Medicago	0	0	4.2	0	6.0
Kochia	7.7	40.0	0	0	2.0
Ephedra	0	0	8.3	94.4	8.0
Erodium	0	0	37.5	16.7	4.0
Lycium	0	0	41.7	22.2	44.0
Gutierrezia	0	0	0	0	0
Lotus	0	0	0	5.6	0
Lupinus	0	0	0	5.6	0
Eriogonum	0	20.0	0	27.8	0
Gilia	0	0	0	0	0
Sida	0	20.0	0	0	0
Unknown flower	7.7	0	0	5.6	4.0
Fungi	76.9	0	0	0	0
Arthropods	76.9	100.0	100.0	77.8	98.0
Bone fragments	0	0	4.2	0	6.0

Schismus arabicus) and the leaves of *Lycium andersonii* and Storksbill (*Erodium cicutarium*). Arthropods were also important (18.2% of the diet) and bone fragments were found in one sample.

Samples collected five weeks later (7–8 July 1988) on the Elkhorn Plain site showed a surprisingly large proportion of leaves in the diet. By far the most important species in the diet was *Ephedra californica* (35.5% relative density, 94% of samples). I also found foliage of *Eriogonum* sp., *Erodium cicutarium*, and *Lycium*

andersonii in relative densities of 1% or greater. I found arthropods in most samples, but they made up only 7.3% of the diet.

Samples collected during from 27–28 May 1989 on the Elkhorn Plain were different from those collected during the same time of year in 1988 (Table 2). Unknown seeds were again the largest category, but arthropods composed a large fraction of the diet (32.5%) and green vegetation composed a small proportion (8.3%) of the material eaten. Only the foliage of *Lycium andersonii* made up

TABLE 2. Mean percentage relative density (see text for explanation) of food types in the diet of the San Joaquin Antelope Squirrel (*Ammospermophilus nelsoni*) for each study site and season. The abbreviation n is the number of individual squirrels from which fecal samples were collected. The mean and standard deviation are given for each food type. The locations of study areas and dates of sample collection are given in the text.

	Paine Spring 1988 n = 13	Paine Summer 1988 n = 5	Elkhorn Spring 1988 n = 24	Elkhorn Summer 1988 n = 18	Elkhorn Spring 1989 n = 50
Seeds					
Grass	22.4 ± 30.7	3.9 ± 5.6	0.5 ± 1.5	9.3 ± 19.4	5.4 ± 12.8
Chenopod	0.8 ± 1.9	6.2 ± 10.8	1.3 ± 6.1	0	0
Composite	35.6 ± 34.6	18.7 ± 33.7	0	1.4 ± 5.9	0.1 ± 0.7
Lycium spp.	0	0	4.8 ± 6.8	22.5 ± 6.1	2.1 ± 5.4
Unknown seed	0	10.3 ± 21.2	49.1 ± 23.9	38.3 ± 32.7	51.9 ± 24.0
Unknown pod	0	0	0	10.7 ± 1.8	0
Leaf Tissue					
Allenrolfea	0.2 ± 0.8	0	0	0	0
Atriplex	6.2 ± 6.6	5.3 ± 10.6	0.6 ± 2.2	0	0
Baileya	0	0.4 ± 1.0	0	0	0
Eremalche	7.2 ± 23.3	0	0.7 ± 9	0	0
Bromus	0	0	4.7 ± 6.2	0.2 ± 0.6	0.1 ± 0.6
Schismus	0	0	10.4 ± 12.4	0	0.5 ± 1.8
Distichlis	0	0	0	0	0.1 ± 0.4
Oryzopsis	0	0	0	0	0.1 ± 0.4
Chrysothamnus	0	0	0	0.2 ± 0.9	0
Trifolium	0	0	0	0	0.1 ± 0.4
Oenothera	2.1 ± 5.7	0	0.1 ± 0.7	0	0.7 ± 0.1
Medicago	0	0	0.1 ± 0.5	0	0.2 ± 0.1
Kochia	0.2 ± 0.5	12.8 ± 8.2	0	0	0.1 ± 1.0
Ephedra	0	0	0.7 ± 2.9	35.5 ± 26.6	0.2 ± 0.9
Erodium	0	0	1.6 ± 2.3	1.0 ± 2.5	0.1 ± 0.6
Lycium	0	0	3.5 ± 5.8	1.0 ± 2.1	6.0 ± 13.9
Gutierrezia	0	0	0	0	0
Lotus	0	0	0	0.2 ± 0.9	0
Lupinus	0	0	0	0.1 ± 0.6	0
Eriogonum	0	0.6 ± 1.4	0	3.4 ± 8.8	0
Gilia	0	0	0	0	0
Sida	0	0.6 ± 1.4	0	0	0
Unknown flower	0.3 ± 1.1	0	0	0.4 ± 1.8	0.1 ± 0.7
Fungi	17.2 ± 19.9	0	0	0	0
Arthropods	7.5 ± 7.8	41.2 ± 21.5	18.2 ± 15.2	7.3 ± 7.6	32.3 ± 20.5
Bone fragments	0	0	0.3 ± 1.4	0	0.2 ± 0.8

more than 1% of the diet, and I found bone fragments in three samples.

DISCUSSION

Hawbecker (1947) studied the diet of San Joaquin Antelope Squirrels at three sites in the Panoche Hills, San Benito County, California, during 1940–1946. Unfortunately, Hawbecker did not present quantitative results (i.e., the number of samples, etc.), the method of sample collection (observation, stomach, or fecal samples) or the method of analysis, but presented qualitative and descriptive results of his sampling, summarized in a figure showing the percentage of various food types in the diet over an annual cycle. He found that the species was omnivorous, eating a variety of plant foods (including leaves, stems, and seeds) as well as arthropods and occasional carrion. The overall

seasonal pattern was one in which green vegetation was most important from December through early May. From early May to December, insects were most important. Seeds were never more than 20% of the relative density of food types but were most important from March to May and from December to January. In 1946, insects were less abundant, and seeds had a higher importance than insects in the summer. Although many plant species were consumed, by far the majority of leaves consumed was Red Brome (*Bromus rubens*) and *Erodium cicutarium*. Both of these plant species are not native to California. A wide variety of arthropods was consumed, including beetles, Jerusalem crickets, June beetles, grasshoppers, and camel crickets. Grasshoppers were the most important summer arthropods, Jerusalem and camel crickets were most important in winter and spring. Seeds important in the diet included *Erodium cicutarium*, *Bromus rubens*, and *Ephedra californica*.

My results differ in some respects from those of Hawbecker (1947), perhaps because of the different habitats at the study sites and study methods. Results of the Hawbecker study came from a population occupying an upland, annual grassland that had been grazed by livestock with virtually no shrub cover. My samples were collected from an *Atriplex*-dominated alkali sink on the floor of the San Joaquin Valley (Paine Preserve) and from an upland, grazed *Ephedra* scrub community (Elkhorn Plain). At the Paine Preserve, seeds were important in both sampling periods, and were at least twice as prevalent as the 20% reported by Hawbecker. I found that fungi were an important spring food, which no author has previously reported being used by this species (Hawbecker 1947; Best et al. 1990); however, the consumption of fungi is widespread among sciurids (Landry 1970). Perhaps winter flooding of the site provides sufficient moisture for spring fungal growth. Leaves remained an important food source into mid-summer, in contrast to the pattern seen by Hawbecker. This probably stems from the availability of perennial shrubs such as *Atriplex*, Iodine bush (*Allenrolfea occidentalis*), and Kochia (*Kochia scoparia*) in combination with a few summer annuals, which did not occur at the Panoche Hills site. Arthropod consumption increased in summer but did not dominate the diet to the extent reported by Hawbecker (1947).

On the Elkhorn Plain, seeds were also important, although the large proportion of unknown seed material makes interpretation of this result difficult. It is possible that seeds were no more important at this site than they were in the study by Hawbecker (1947). Leaves were consumed during all three sample periods on the Elkhorn Plain, and included the foliage of grasses and shrubs, with a few annual herbs represented in smaller quantities. The foliage of perennial shrubs, particularly *Ephedra*, *Lycium*, and *Eriogonum* was an important source of green vegetation not available on the study area of Hawbecker in any significant quantity. The heavy consumption of *Ephedra* foliage during the sampling period in summer

1988 sampling period was a striking deviation from the pattern described by Hawbecker. The small proportion of arthropod material in the same sample is also quite different from the heavy preponderance of arthropods in the summer samples of Hawbecker.

The results of my study demonstrate a broader array of food types used, both generally and specifically, in comparison with the results from the Panoche Hills. This is not surprising, given that both of my study areas had more complex and diverse vegetation than the shrubless annual grassland sites of Hawbecker. The ecologically similar White-tailed Antelope Squirrel (*Ammospermophilus leucurus*) has been studied more extensively and has been shown to be omnivorous, consuming leaves, seeds, arthropods, and other materials (Belk and Smith 1991). Green vegetation is the primary food from December through May, seeds are consumed at all times of year, and arthropods are eaten primarily in late summer and autumn (Belk and Smith 1991).

There are significant limitations to interpretation of the data presented here. I collected samples incidental to other work, and as a result, the time period in which samples were collected was limited for the most part to early and mid-summer (late May to mid-July). In addition, the two years of study took place after relatively low winter rainfall. It is likely that earlier sampling at these sites would reveal more use of herbaceous green vegetation. Further diet studies should include a broader range of seasons and annual conditions and should include estimates of the availability of food types. Studies that compare food availability to reproduction would be especially useful.

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Solar Energy Development and Endangered Species in the San Joaquin Valley, California: Identification of Conflict Zones

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Abstract.—A number of animal and plant species in the San Joaquin Valley (SJV) of California are rare due to profound habitat loss and degradation. A significant portion of the remaining habitat for these species also has high potential for solar energy generation. We conducted a spatially explicit GIS analysis of lands in the SJV to identify areas of potential conflict between rare species and solar energy development and also to identify areas where such conflict would be minimized. We modeled solar energy generation potential and also modeled habitat suitability for five federally listed animal species whose ranges encompass those of additional rare species. We then layered the model results to identify areas of greater or lesser conflict. Approximately 4,145 km^2 have moderate to high potential for solar energy development and also have moderate to high quality habitat for listed species. The potential for environmental conflicts is high on these lands. Approximately 8,436 km^2 have moderate to high potential for solar energy development but no or low-quality habitat for rare species. These lands are the optimal sites for solar energy generation projects. Furthermore, siting projects on lands with no or marginal habitat value could enhance the value of these lands for rare species and create linkages between occupied areas. Our approach can be applied in other locations to identify general areas and even specific locations where siting solar facilities would result in minimal or no impacts to sensitive resources and possibly even enhance regional conservation efforts.

Key Words.—*Ammospermophilus nelsoni*; forbs; grasses; San Joaquin Desert; shrubs

Introduction

Solar power is a rapidly growing renewable energy source world-wide, and concomitant with this has been an accelerated rate of construction of utility-scale solar energy generation facilities. The marked increase in such facilities has been particularly acute in California (Solar Energy Industries Association 2016) where optimal conditions (e.g., flat terrain, high insolation rates) are abundant, and where the state legislature passed a bill in 2015 requiring all power-supplying utilities to obtain at least 50% of their electricity from renewable energy sources by 2030 (de León 2015). This bill was followed by another in 2018 (de León 2018) that mandated that the 50% target be reached by 2026, that 60% be achieved by 2030, and that renewable and zero-carbon sources supply 100% of retail sales of electricity by 2045. This could further accelerate the construction of solar facilities in California.

Although the rapid proliferation of solar facilities is positive with regards to helping ameliorate climate change impacts, a significant concern is adverse effects to sensitive biological resources resulting from these facilities (Lovich and Ennen 2011; Turney and Fthenakis 2011, Hernandez et al. 2015), particularly when the facilities are constructed on lands that provide habitat for species at risk (Leitner 2009; Lovich and Ennen 2011; Stoms et al. 2013; Moore-O'Leary et al. 2017). Some of the rare species affected by recent solar projects in California include the San Joaquin Kit Fox (*Vulpes macrotis mutica*), federally listed Endangered (FE) and

California-listed Threatened (CT), Giant Kangaroo Rat (*Dipodomys ingens*), FE and California-listed Endangered (CE), Desert Tortoise (*Gopherus agassizii*), Federally listed Threatened (FT) and CT, and Mohave Ground Squirrel (*Xerospermophilus mojavensis*), CT (Leitner 2009; Moore-O'Leary et al. 2017).

Numerous utility-scale solar facilities have been constructed or proposed for the San Joaquin Valley (SJV) region of California. In addition to high insolation rates and an abundance of flat terrain, relatively low land prices and proximity to transmission corridors enhance the attractiveness of this region for such facilities (Pearce et al. 2016); however, a large number of rare species also occurs in this region due to geographic isolation and high levels of endemism coupled with profound habitat loss (U.S. Fish and Wildlife Service [USFWS] 1998). By 2004, approximately 70% of the over 3.9 million ha of historical habitat in the SJV had been replaced by irrigated agriculture and urban development (Kelly et al. 2005). Thus, developments in the remaining natural lands further enhance the risk of extinction for multiple animal and plant species.

We conducted a spatially explicit analysis using a GIS-based model to assess location-specific potential for conflicts between listed species and solar energy development in the SJV. Our objectives were to identify areas more conducive to solar energy facilities due to high solar energy potential and low impacts to rare species, and identify areas where solar projects should be avoided based on the presence of high-value habitat and the potential for the occurrence of multiple rare species.
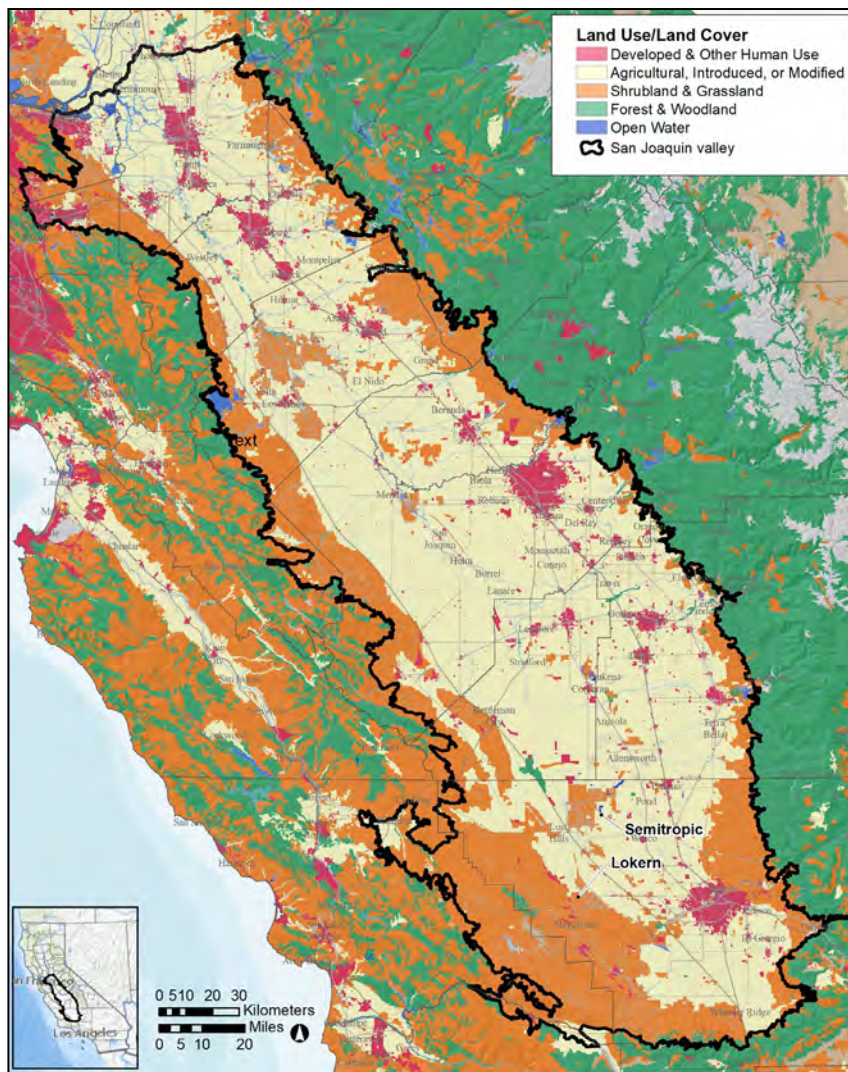
FIGURE 1. The San Joaquin Valley region in California. Land use/land cover classes are from a state-wide vegetation layer (University of California-Santa Barbara Biogeography Lab 1998) combined with a more recent layer of farmland and urban areas in California (California Department of Conservation 2015).

This approach can be applied in other regions as well where the potential for conflict between rare species and solar energy development is high.

METHODS

Study area.—The SJV in central California (Fig. 1) extends about 415 km from north to south, and encompasses approximately 3.44 million hectares below the 152-m (500-ft) contour (USFWS 1998). The SJV is bounded on the east by the Sierra Nevada, on the west by the Coast Ranges, on the south by the Transverse Ranges and on the north by the extensive delta of the Sacramento and San Joaquin Rivers. The SJV is an arid region characterized by hot, dry summers and cool, rainy winters. Historical habitat types included arid grasslands, arid shrublands, woodland savannahs, and lakes and marshes on the valley floor connected by rivers and sloughs (USFWS 1998, Germano et al. 2011). The savannahs, lakes, and wetlands have been all but eliminated by agricultural and urban development, and

the grasslands and shrublands have been significantly reduced to a fraction of their former acreage (USFWS 1998; Kelly et al. 2005). Urban regions in the SJV are growing rapidly and major population centers include Stockton, Modesto, Merced, Fresno, Visalia, and Bakersfield. Most constructed and planned solar energy plants are located in the more arid western and southern SJV described by Germano et al. (2011) as the San Joaquin Desert.

To examine conflicts between listed species and energy development in the SJV, we developed a GIS-based model (see Appendix A for model schematic) to determine how those areas best-suited for solar development compare with the suitability of remaining habitat for five federally or state listed animal species typically associated with the San Joaquin Desert. The five species were the Blunt-Nosed Leopard Lizard (*Gambelia sila*), FE and CE, San Joaquin Kit Fox, FE and CT, San Joaquin Antelope Squirrel (*Ammospermophilus nelson*), Federal Species of Concern and CT, Giant Kangaroo Rat, FE and CE, and San Joaquin Kangaroo Rat (*Dipodomys nitratoides*),

TABLE 1. Criteria used to evaluate suitability for large-scale solar development in the San Joaquin Valley, California. Slope was averaged over a 128-ha (320-ac) neighborhood.

Criteria	Potential for Solar Development		
	None to low	Moderate	Highest
Land use	Developed (urban areas, industrial, extractive), permanent crops (orchards or vineyards), open water, forests, or wetlands.	Irrigated farmland excluding permanent crops (e.g., row crops)	Rangeland, fallow/idle farmland, or dryland-farmed areas (e.g., winter wheat)
Slope	> 15°	< 15°	< 15°
Protected lands	Protected lands (public lands, private conservation lands, or conservation easements)	Other private land	Other private land
Insolation	N/A	5.68 - 6 kWh/m^2/day (or row crops with > 6 kWh/m^2/day)	6.00–6.42 kWh/m^2/day

which consists of three subspecies: Tipton Kangaroo Rat (*D. n. nitratoides)*, FE and CE, Fresno Kangaroo Rat (*D. n. exilis*), FE and CE, and Short-nosed Kangaroo Rat (*D. n. brevinasus*), Federal Species of Concern and California Species of Special Concern. We selected these species because of their relatively wide distributions, which encompass those of most other rare species occurring in the San Joaquin Desert. GIS models have been used elsewhere to identify areas of conflict between solar energy development and conservation goals (Cameron et al. 2012; Stoms et al. 2013). Our analysis did not explicitly include other regionally important components of conservation concern, in particular wetland habitats and associated species, and listed or rare plants.

Suitability for solar development.—We evaluated suitability for solar development using methods similar to those used by Butterfield et al. (2013) to evaluate site-suitability for large-scale (e.g., photovoltaic sites > 20 MW) solar facilities. Our criteria included land use, terrain, protected land status, and insolation rates (Table 1). These criteria are not comprehensive and other factors, such as proximity to transmission corridors and land values, also can affect site selection for solar farms; however, as noted by Pearce et al. (2016), these other factors can change rapidly and so we did not consider them in this analysis. We assumed that utility-scale solar facilities sites would need to be larger than 80 ha (200 acres) in area based on a high estimate (75th percentile) of acres/MW for photovoltaic solar sites larger than 20 MW estimated by the National Renewable Energy Laboratory (NREL 2013), and screened out areas smaller than that minimum size. Because we did not include all possible factors, some areas identified as suitable may be impractical to develop because of other limiting factors.

We developed a GIS layer of current land use classes based on a combination of the National Agricultural Statistics Service (NASS) 2014 cropland data layer (U.S. Department of Agriculture NASS 2015) and the California Department of Conservation (CDOC) Farmland Mapping and Monitoring Program (FMMP) 2012 important farmland layer (CDOC 2015). We

combined land use classes from both layers to create a simplified classification (Table 2) that we used to evaluate both solar site potential and habitat availability. The two source layers (FMMP, NASS) are created using different methods and for different purposes and so differ in thematic accuracy (correct classification) and thematic resolution (number of mapped land use classes). The FMMP layer is created using direct interpretation from aerial photography and field observations (CDOC 2004), whereas NASS uses semi-automatic classification of satellite imagery. Based on a comparison of NASS classifications to observed classifications in reconnaissance surveys (Endangered Species Recovery Program, unpubl. data), we found that semi-automatic classification techniques are less reliable for land uses that have similar vegetation and ground cover such as rangeland and idle farmland (two important categories for our analysis). We also found that the FMMP included a more accurate depiction of the extent of rangeland but lacked the thematic resolution (detailed land use categories) of NASS (e.g., orchards, vineyards, wetlands, and forest). Because it takes less time to produce, NASS is updated on a yearly cycle, and is usually more current than what is available from FMMP at any given time. To

TABLE 2. Land use classification used to evaluate solar and habitat potential in the San Joaquin Valley, California. The acronym FMMP = Farmland Mapping and Monitoring Program 2012 important farmland layer from the California Department of Conservation and NASS = National Agricultural Statistics Service 2014 cropland data layer. For primary source, NASS[1] has no equivalent category in FMMP.

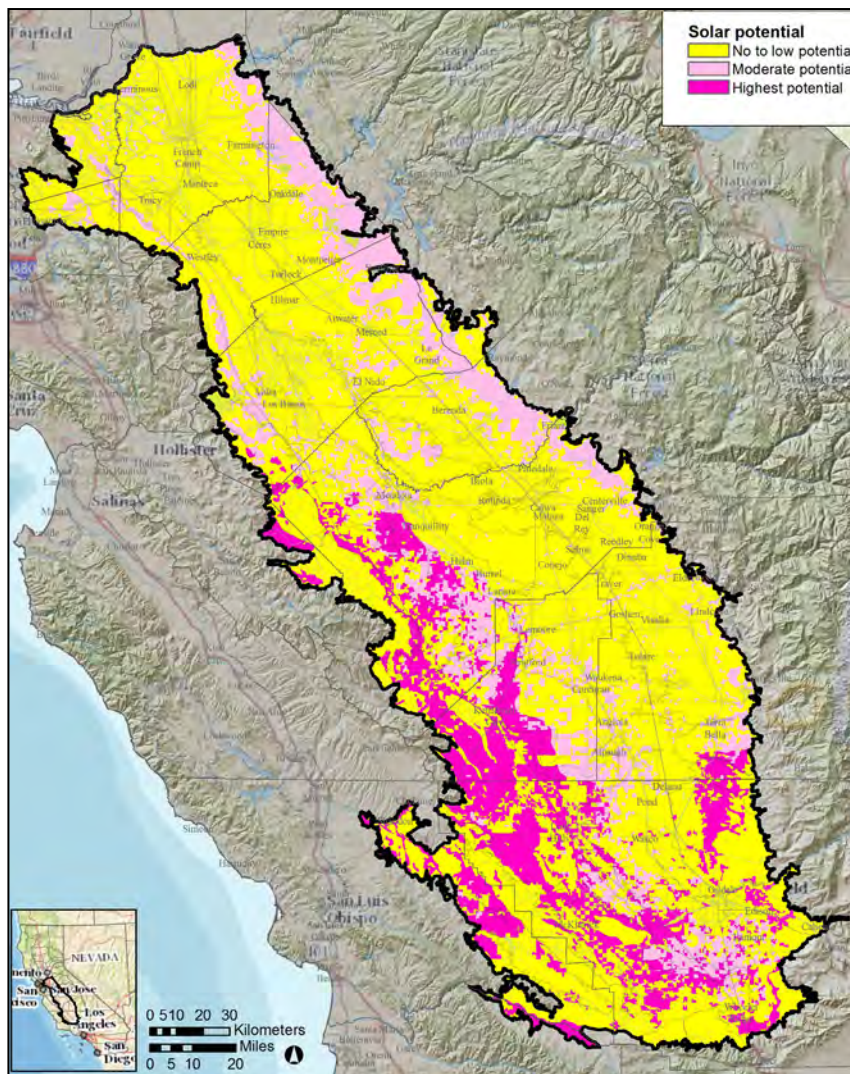| Land use class	Primary source	Secondary source
Urban/Industrial/Other developed	FMMP	NASS
Permanent crops	NASS[1]	-
Row crops	NASS[1]	-
Fallow or dryland-farmed	NASS[1]	-
Rangeland	FMMP	NASS
Barren	NASS[1]	-
Forests or wetlands	NASS[1]	-
Water	FMMP	NASS

Figure 2. Estimated solar potential based on land use, protected land status, slope, and insolation in the San Joaquin Valley, California.

take advantage of both the thematic accuracy of FMMP and thematic resolution of NASS, we used a GIS overlay analysis to combine information data from both sources using the following classification rules: Where FMMP land use was classified as agricultural land or unknown, we used the more-detailed categories from the NASS. Otherwise, we used the FMMP land use categories that we found to be more-thematically accurate for non-agricultural areas, urban areas, and water. For the non-agricultural area (e.g., rangeland), we added supplemental information where the more-detailed NASS data had identified areas of forest or wetlands (classes included in the NASS but not included in the FMMP).

We calculated slope (in degrees) from digital elevation models available from the U.S. Geological Survey (USGS) National Elevation Program (USGS 2014). To screen out small patches of flat slope in otherwise steep terrain, we used a spatial averaging function (Focal Statistics in ArcGIS). Specifically, we calculated each cell as the mean value of cells within a 640-m-radius circular area (approximately 320 ac or 128 ha).

We screened out areas identified as protected fee or easement lands (GreenInfo Network 2015). While fee and easement lands have varying levels of protection from development such as large-scale utility solar facilities, we considered them all as having generally higher protection against solar development and focused our analysis on private lands. We also estimated insolation using solar resource data available from the National Renewable Energy Laboratory (NREL 2012). Solar resource data were derived from NREL estimates for photovoltaic energy (tilt = latitude collector) available as 10-km grids. To match the resolution of our other data sources, we converted the grids to a higher-resolution surface using a spatial interpolation function using an Inverse Distance Weighting function in ArcGIS (Power = 2; Search Radius = 12 neighboring cells). We combined map layers for the four criteria using a series of GIS Map Algebra steps statements (Appendix A) to create a composite map of potential suitability for solar development consisting of three categories: Low, Moderate, and High (Fig. 2).
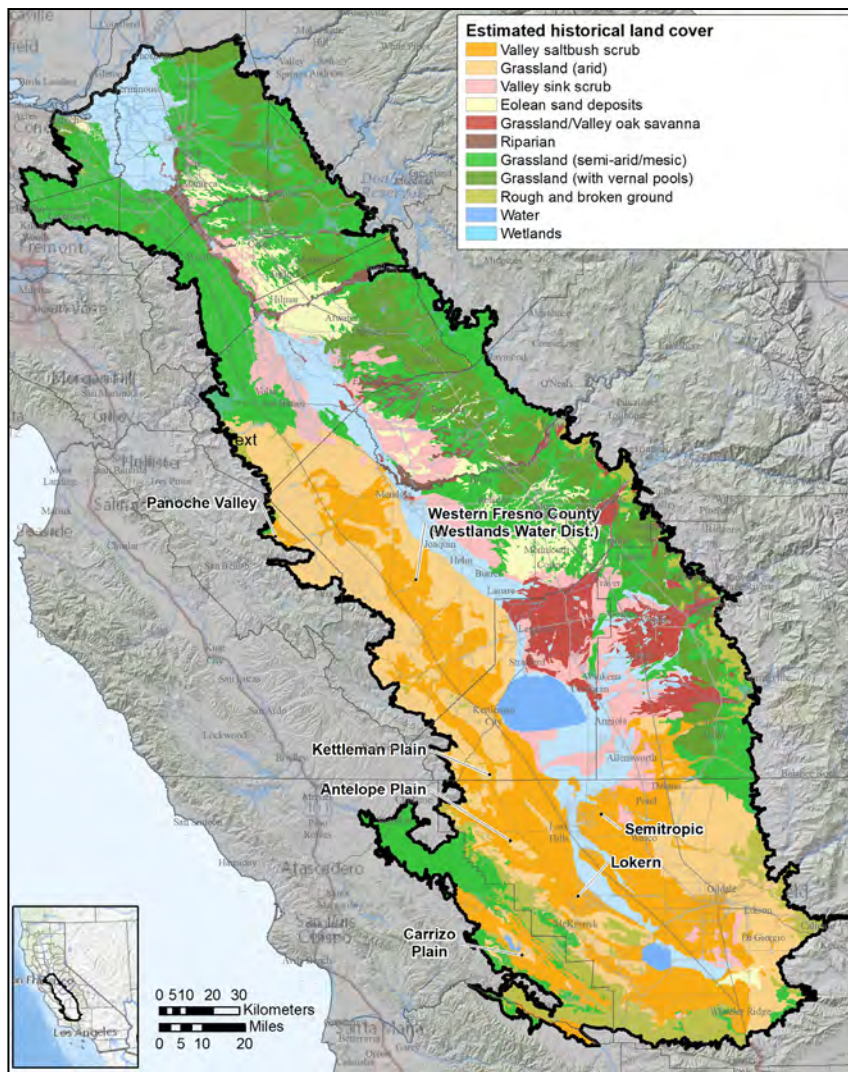
Figure 3. Estimated historical land cover in the San Joaquin Valley, California.

Suitability for listed species.—We evaluated habitat quality for the listed species using an approach similar to Germano et al. (2011) who used the distribution of multiple species along with ancillary information to identify a general region (i.e., San Joaquin Desert) important to multiple arid-adapted species of the SJV. Our approach was to develop a relatively detailed (approximately 1:125,000) GIS layer of historical land cover. To do this (Fig. 3), we digitized map units from a set of soil surveys of the San Joaquin Valley that pre-date most of the conversion of rangelands to irrigated agriculture (Holmes et al. 1919; Nelson et al. 1918; Nelson et al. 1921). To fill some data gaps near the edges of our study area, we also used information from contemporary soil surveys (U.S. Department of Agriculture, Natural Resources Conservation Service 2014, 2015). We assigned vegetation classes to map units primarily using descriptions (and example photographs) of soil series map units (Appendix B). For example, series descriptions may include descriptions of grazing conditions, presence of brush, or information on terrain

and drainage.

While we assigned classes mostly by the description of the soil type, we also reviewed historical map sources (Hall, W.H. 1890. Topographic and irrigation maps of San Joaquin Valley, Sheets 1-4. Water Resources Archives, University of California, Berkeley, California; Piemeisel and Lawson 1937; Kuchler 1977; Werschull et al. 1984), historical photographs (MVZ 2015), and climate data (PRISM Group at Oregon State University 2014) and in some cases updated our classification based on climate or another secondary source (Appendix B). We used a subset of species occurrence records from the Natural Diversity Database (CDFW 2014) and the recovery plan for upland species of the San Joaquin Valley of California (USFWS 1998) along with habitat descriptions from literature sources (Grinnell 1918, 1922, 1932) to assign historical presence of each of our target species to our historical land cover map units. We used the occurrence records and habitat descriptions to estimate the historical geographic distribution of each species and then used the historical vegetation classes to refine the distribution. For
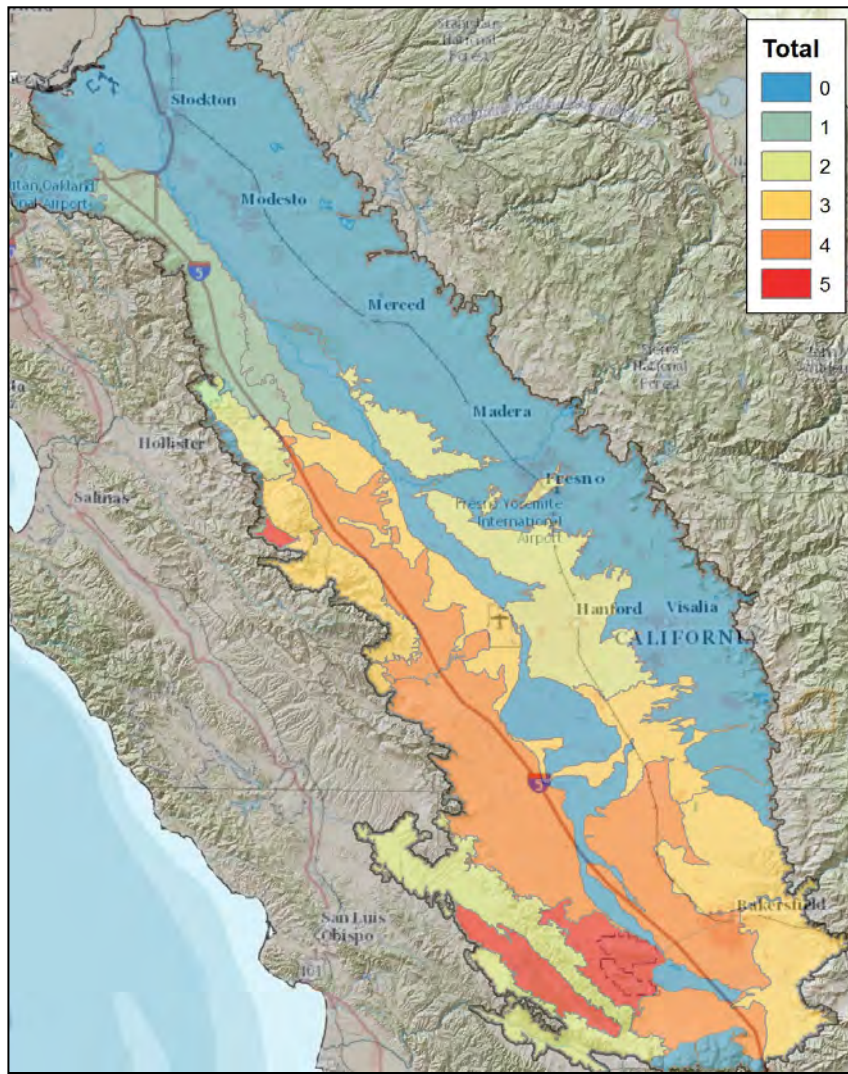
Figure 4. Total number of range overlaps for Giant Kangaroo Rats (*Dipodomys ingens*), Short-nosed Kangaroo Rats (*D. nitratoides brevinasus*), Fresno Kangaroo Rats (*D. n. exils*), Tipton Kangaroo Rats (*D. n. nitratoides*), San Joaquin Antelope Squirrels (*Ammospermophilus nelsoni*), Blunt-nosed Leopard Lizards (*Gambelia sila*), and San Joaquin Kit Foxes (*Vulpes macrotis mutica*) based on historical ranges in the San Joaquin Valley, California.

example, we used historical records of Fresno Kangaroo Rats to identify map units where they were present, but also included contiguous or nearby map units with similar conditions. For map units with few occurrence records, we reviewed the descriptions and sources of the record to screen out those with high spatial uncertainty or those where the species identification was questionable (e.g., San Joaquin Kit Fox records based only on presence of sign but no captures).

We estimated historical habitat value by adding up the number of co-occurring species (Fig. 4). Using the slope

layer, we identified and removed steep and rugged lands (> 30° slope) and grouped the remaining lands into four habitat quality categories: No to low, Low to moderate, Moderate to high, and Highest (Fig. 5). We combined the estimated composite historical habitat layer (Fig. 5-A) with a layer of contemporary land use (Fig. 5-B). Contemporary rangelands (e.g., grasslands, saltbush scrub) were assigned their estimated historical value and non-rangelands (e.g., irrigated farmland, developed areas) were assigned a value of No to Low habitat value (Table 3, Fig. 6).

Table 3. Criteria used to evaluate habitat quality for potential solar projects in the San Joaquin Valley, California. Slope was averaged over a 128-ha (320-ac) neighborhood.

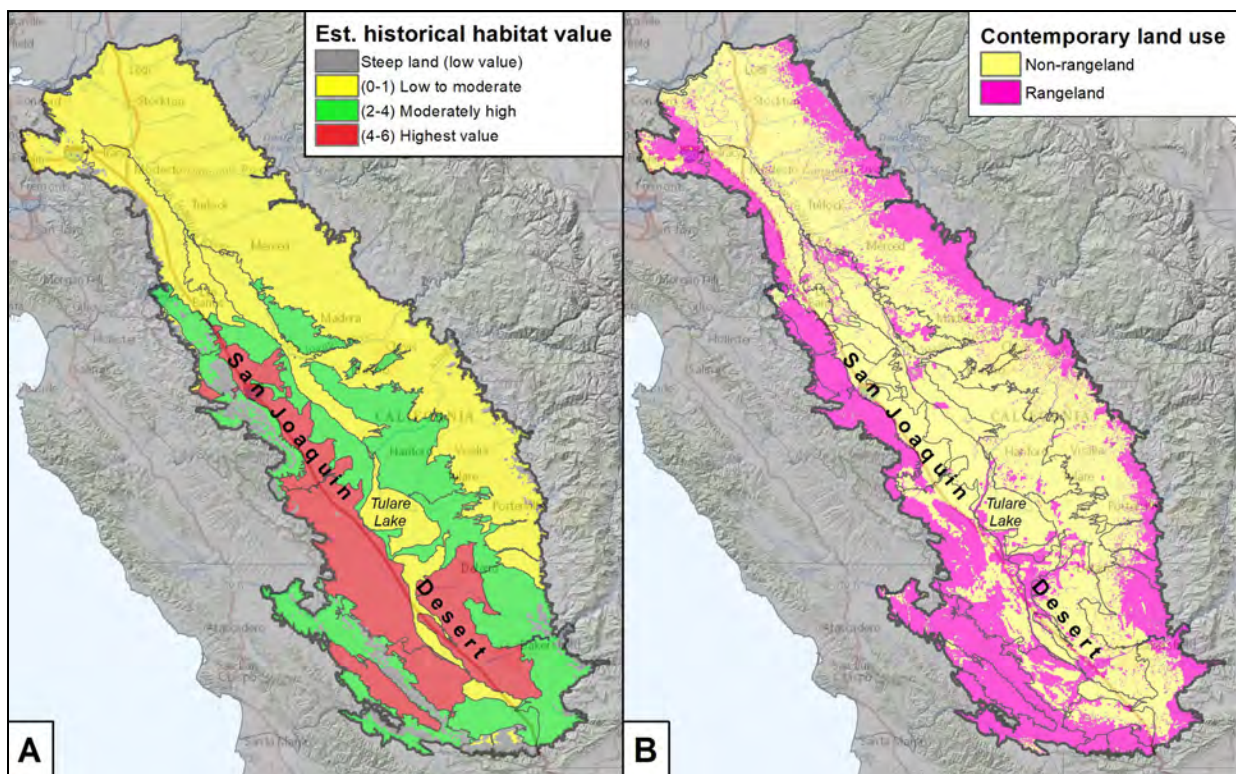| Criteria	Habitat value			
	None to low	Low to moderate	Moderately high	Highest
Estimated historical species ranges	-	0–1 overlapping range	2–4 overlapping ranges	Greater than 4 overlapping ranges
Land use	Not rangeland	Rangeland	Rangeland	Rangeland
Slope	> 30°	< 30°	< 35°	< 30°

Figure 5. Estimated historical habitat value (A) and contemporary land use showing current rangeland or non-rangeland (B) in the San Joaquin Valley, California.

Results

We determined the habitat value for listed species and the potential for solar development for 42,707 km² in the SJV (Table 4). In particular, arid shrublands and grasslands tended to have attributes favorable for solar energy development. Thus, of the species we evaluated, Blunt-nosed Leopard Lizards, San Joaquin Antelope Squirrels, Giant Kangaroo Rats, Short-nosed Kangaroo Rats, and San Joaquin Kit Foxes would be most affected. Tipton and Fresno kangaroo rats primarily occur in alkali sink habitat, which was less suitable for solar development and so would be less affected.

Nearly 40% of areas with the highest potential for solar development were in areas with the highest habitat value. Thus, these areas can be considered conflict zones. This overlap increased to 64% when both the highest and moderate to high habitat value categories were considered (Table 4; Fig. 7). These conflict areas were concentrated in the southwestern portion of the SJV. Nearly a third (31%) of areas with the highest potential for solar development were in areas of less conflict (e.g., No to Low quality habitat consisting of marginal or idle farmland). Likewise, two thirds (67%) of the areas of highest habitat value were in the areas with the highest potential for solar development (Table 4; Fig. 7).

Table 4. Cross-tabulation of area for zones of suitability for solar development and habitat quality zones in the San Joaquin Valley, California.

| Habitat Value	Solar Potential			Total
	Low	Moderate	High	
None to low value	24,821 km² (9,584 mi²)	3,002 km² (1,159 mi²)	1,789 km² (691 mi²)	29,612 km² (11,433 mi²)
Low to moderate value	1,931 km² (746 mi²)	3,337 km² (1,288 mi²)	308 km² (119 mi²)	5,576 km² (2,153 mi²)
Moderate to high value	2,349 km² (907 mi²)	440 km² (170 mi²)	1,375 km² (531 mi²)	4,164 km² (1,608 mi²)
Highest value	1,025 km² (396 mi²)	85 km² (33 mi²)	2,245 km² (867 mi²)	3,355 km² (1,295 mi²)
Total	30,126 km² (11,632 mi²)	6,864 km² (2,650 mi²)	5,717 km² (2,207 mi²)	42,707 km² (16,489 mi²)
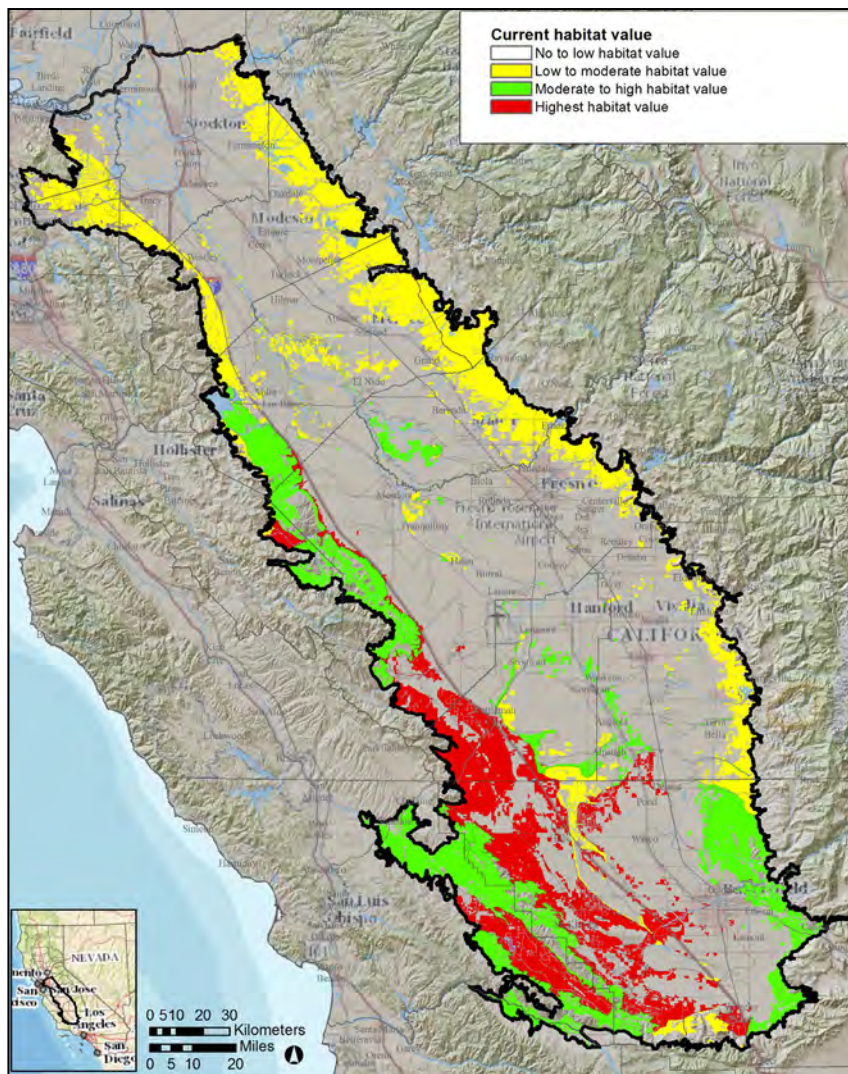
FIGURE 6. Estimated habitat value based on historical species ranges, land use, and slope in the San Joaquin Valley, California.

DISCUSSION

Our analysis indicated that there is considerable overlap between site qualities needed for solar energy generation and those that constitute suitable habitat for listed species in the SJV. Consequently, a large proportion of the remaining high-quality habitat for these species also is optimal for solar energy development. This overlap results in the potential for significant conflict between development of new energy sources and conservation of at-risk species. Most historical habitat for these species has been converted to other land uses (e.g., agriculture) and habitat loss continues to be the greatest threat to listed arid-adapted species (USFWS 1998). Additional conversion of habitat for any reason, including solar energy development, could further imperil these species. Furthermore, although our analysis was based on select species, a number of other rare species share similar habitat requirements with the featured species (USFWS 1998), and therefore the results of our analyses are applicable to a large suite of species of conservation concern in the SJV.

Based on our analyses, there are approximately 4,145 km² (1,601 mi²) with moderate to high potential for solar energy development and that also constitute moderate to high quality habitat for listed species. These lands comprise the highest potential for conflict. Securing permits to develop these lands, particularly from agencies such as USFWS and California Department of Fish and Wildlife that are charged with protecting listed species, is difficult and also costly due to the complex impact analyses and substantial mitigation measures typically required. Furthermore, environmental groups commonly have filed lawsuits against project proponents proposing solar energy projects in good quality habitat and this further increases the cost of constructing solar facilities. For three large solar farms recently constructed in high quality habitat for listed species in the SJV (Topaz Solar Farms, California Valley Solar Ranch, Panoche Valley Solar Farm), impact analyses and permitting required several years to complete and mitigation costs were in the 10s of millions of dollars (David Hacker, pers. comm.). Environmental groups initiated legal challenges to all three projects resulting in many more millions of dollars
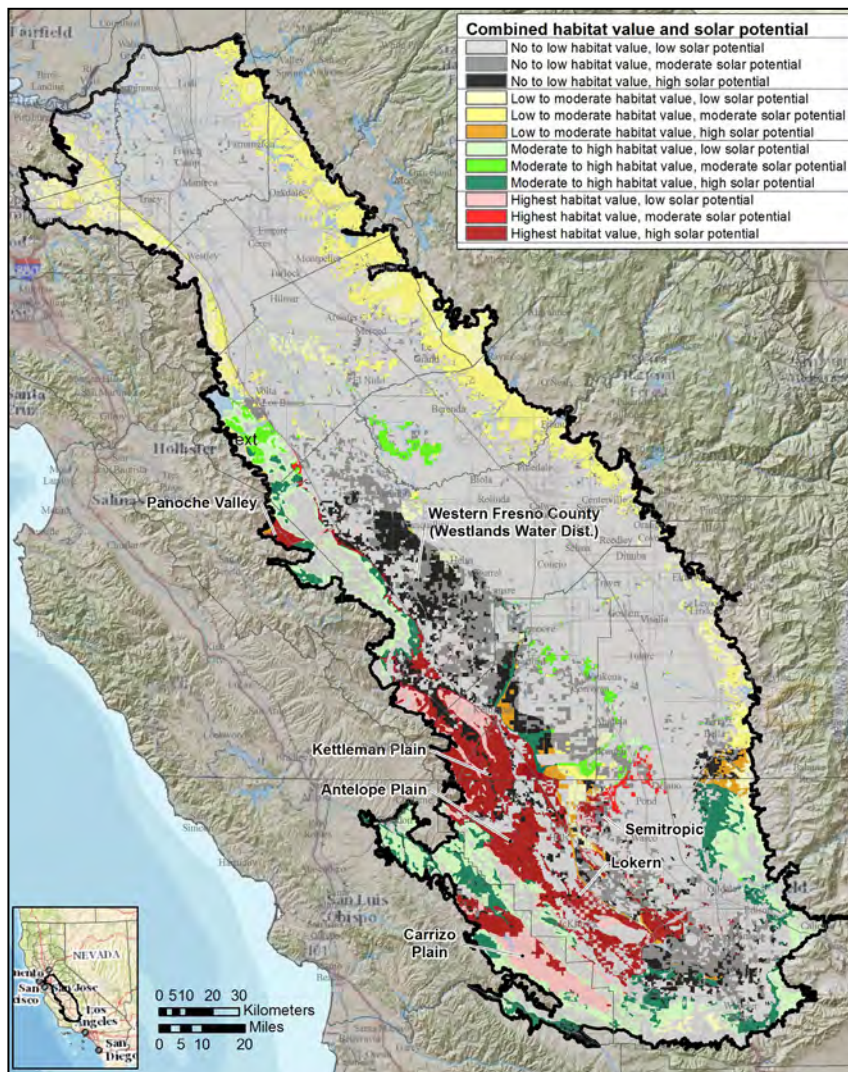
Figure 7. Combined suitability for solar development with contemporary habitat conditions for five listed animal species in the San Joaquin Valley, California.

in additional mitigation costs.

Potential conflict areas with moderate to high habitat value and moderate to high potential for solar energy development are particularly concentrated in the southwestern portion of the SJV from Kern County up into southwestern Fresno County. Other areas include private lands in the northern and eastern Carrizo Plain, valley floor lands in northern Kern and southern Tulare counties, and the Panoche Valley region in eastern San Benito County. These areas all are recognized as being important for the conservation and recovery of the listed species considered in this report and other rare species as well (USFWS 1998).

Conversely, approximately 8,436 km^2 (3,257 mi^2) have moderate to high potential for solar energy development but no to moderate value quality habitat for listed species. These lands constitute more optimal sites for solar energy generation projects. Conflicts with listed species would be minimal or non-existent on these lands. Permit acquisition would be easier and mitigation requirements would be lower. With the ample availability of lands that have high potential for solar development but low habitat

value for listed species, there appears to be abundant opportunity to site new solar projects in areas where at-risk species will be minimally affected, and could reduce the additional costs (e.g., mitigation requirements) associated with higher-quality habitats.

Lands with low habitat value but high potential for solar energy development are scattered throughout the southern SJV with particular concentrations in western Fresno County, southern Kings County, and southern Kern County. There also is a small concentration of such lands on the east side of the valley on the Kern-Tulare County boundary. Many of the lands in western Fresno County are in the Westlands Water District (https://wwd.ca.gov/) where considerable agricultural land already has been taken out of production (retired) or otherwise retired due to salt concentrations and drainage issues (Brian Cypher et al., unpubl. report). Solar energy generation would constitute an excellent alternate use of these lands.

GIS-based approaches have been used previously to identify areas of conflict with solar energy development. Cameron et al. (2012) used GIS modeling to identify areas of least conflict between biodiversity values and

solar energy development in the Mojave Desert of California and Nevada. Stoms et al. (2013) conducted a similar effort for the Mojave and Colorado Desert regions in California. Three previous analyses have been conducted for the SJV, although the objectives, methods, and conclusions differed from ours. Butterfield et al. (2013) identified 1,592 km^2 (615 mi^2) of lands where suitability for solar farms was high but conservation value was low. Similarly, Jane Cowan et al. (unpubl. report) and Pearce et al. (2016) identified 4,047 km^2 (1,563 mi^2) and 1,902 km^2 (734 mi^2) respectively, of lands where conflict between solar development and conservation would be limited. These studies and ours differed in two significant ways. A much broader collection of lands (e.g., fallow agricultural lands, wetlands), were used to assess conservation values, but in many instances these lands do not support listed species. We focused on lands considered important for arid-adapted listed species (USFWS 1998). Each of these other analyses also included conservation of agricultural lands as an objective, and this likely is the reason that the least conflict lands totaled considerably less than the 8,436 km^2 (3,257 mi^2) that we identified. Agricultural lands in the SJV generally are flat and therefore are optimal for solar farms, but agricultural lands have little or no value for the listed species we considered (Warrick et al. 2007; Cypher et al. 2013).

A notable point of agreement among our study and the three others conducted in the SJV is the identification of a sizeable concentration of lands in western Fresno County where conflicts between solar energy development and conservation values would be minimal. As described previously, many of these lands lie within the Westlands Water District. Due to soil salinity and other issues, many acres within the District have been taken out of agricultural production. Thus, this region potentially could serve as a focal area for solar energy production.

Siting projects in areas with no or marginal habitat value actually might increase the value of these lands for listed species. Preliminary data from recently constructed solar generating facilities indicated continued, and in some cases increased, use by listed species (Cypher et al. 2019). The Topaz Solar Farms in northeastern San Luis Obispo County was largely constructed on active and fallowed dry-land farmed fields. San Joaquin Kit Foxes were present in low abundance on the site prior to construction, and they continue to occupy the site now that construction has been completed and the facility is fully operational (Meade, Althouse and Meade, Inc., pers. comm.). The results of surveys involving genetic analyses of fecal samples indicate that kit fox numbers have increased on the site (Jesus Maldonado and Tammy Wilbert, unpubl. report). Similarly, kit foxes continue to use another nearby solar facility, the California Valley Solar Ranch (Robyn Powers, pers. comm.). This facility was constructed on lands that were previously farmed or intensively grazed. Both solar sites appear to be used by kit foxes to fulfill all life-history requirements (e.g., foraging, denning, resting). Reproduction by kit foxes also has been documented on both sites. Furthermore, Giant Kangaroo Rats were present in low numbers on the California Valley Solar Ranch lands prior to construction and continue to be present and have even increased in some areas now that construction has been completed (Robyn Powers, pers. comm.). Conservation measures that have facilitated use of these solar facilities by listed species include permeable fencing, movement corridors, vegetation management, enhancements such as artificial dens, and prohibition of rodenticide use.

The examples above indicate that if designed and managed appropriately, solar generating facilities can provide habitat value for listed species. Given the overlap in habitat requirements (USFWS 1998) among the listed species used in our analyses, we predict that San Joaquin Kangaroo Rats, San Joaquin Antelope Squirrels, and Blunt-Nosed Leopard Lizards also potentially would use solar facilities, similar to that observed for San Joaquin Kit Foxes and Giant Kangaroo Rats. Thus, solar facilities constructed in low value habitat adjacent to lands occupied by any of these species might actually increase the amount and patch size of useable habitat. Such construction of solar facilities could be particularly valuable if sited in such a manner as to create a corridor across marginal habitat to link areas of higher quality habitat. With the extensive fragmentation of habitat that currently exists in the SJV ecoregion (e.g., USFWS 1998; Kelly et al. 2005; Cypher et al. 2013), the potential for improving conditions for listed species by connecting habitat patches is immense. The recovery plan for upland species in the SJV (USFWS 1998) specifically calls for establishing corridors and improving connectivity in the region in western Fresno County that includes the Westlands Water District. As described previously, species habitat values are generally low and solar energy development potential is relatively high in this region, and solar projects potentially could contribute to conservation strategies as well-managed solar facilities could provide greater habitat value than the existing agriculture.

Per our previous caution, our analysis did not consider all possible factors that could influence the selection of a proposed site for a solar facility in the SJV (for example, listed species associate with wetlands). Our analysis, however, constitutes a useful decision support tool for identifying general areas and even specific locations in this region where siting such facilities would result in minimal or no impacts to listed species. Of the remaining 7,519 km^2 of moderate and high-quality habitat in the SJV, almost half (48.2%) is also highly suitable for solar energy generation. Consequently, the potential for conflict between these two competing land uses is high; however, there are 29,612 km^2 of low-quality habitat that also is highly suitable for solar energy generation. Thus, there is abundant opportunity to site solar energy plants on lands that will not adversely affect listed species.

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Appendix A. Geographic Information System (GIS) model for assessing overlap between solar energy potential and habitat suitability for five at-risk animal species in the San Joaquin Valley, California.

Solar Potential (0-2)

Habitat Value (0-3)

Land Use

1 = Urban/Industrial/Other developed
2 = Permanent crops
3 = Row crops
4 = Fallow or dryland-farmed
5 = Rangeland
6 = Barren
7 = Forested/Wetland
8 = Water
9 = Other

Combined (HV*10+SP)
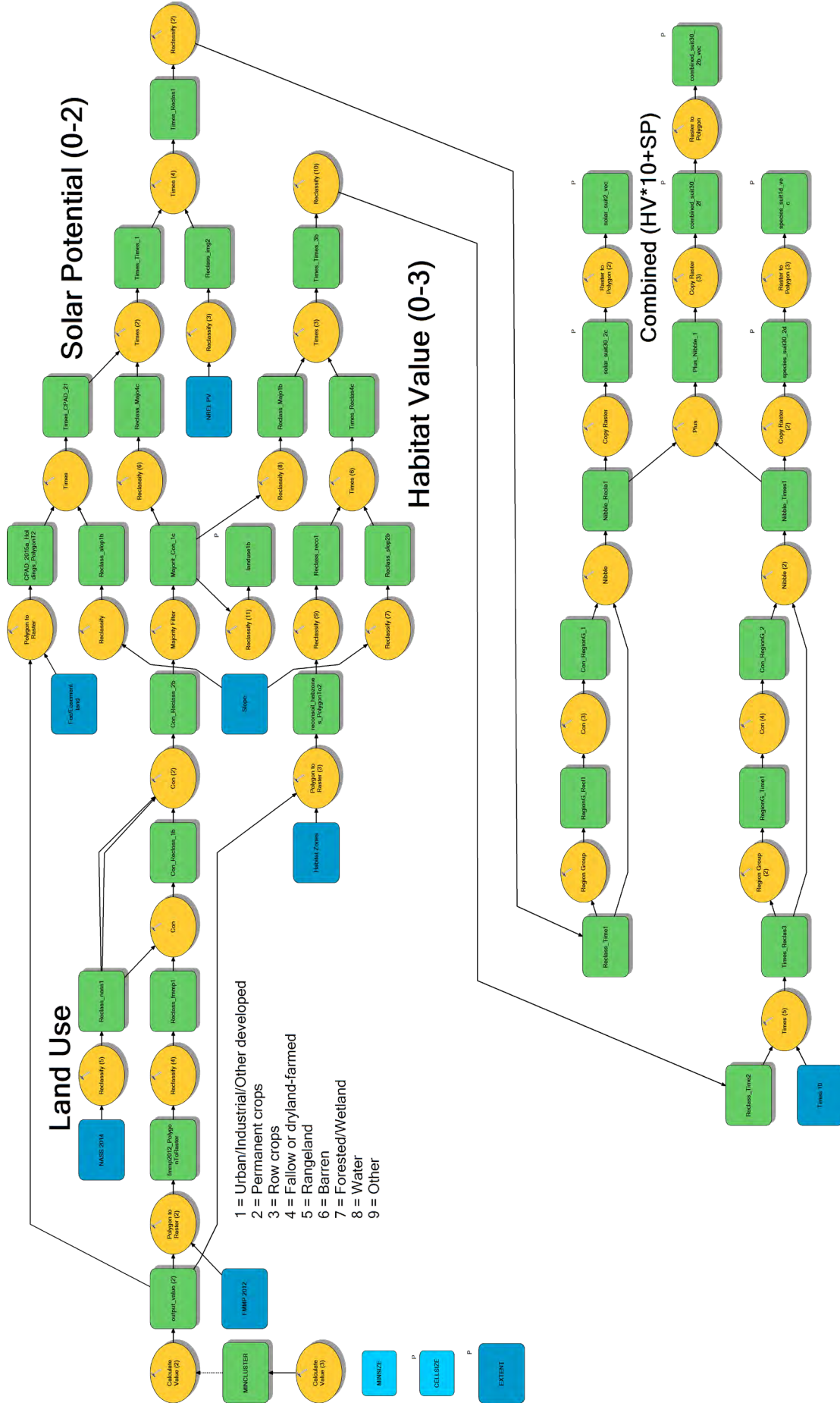
Appendix B. Soil series and vegetation classifications for historical land cover.

Soil series	Vegetation class	Primary classification source	N map units
Aiken loam	Grassland	Soil description	11
Aiken stony loam	Grassland	Soil description	25
Alamo clay adobe	Grassland	Soil description	45
Altamont adobe soils	Grassland	Soil description	33
Altamont and Diablo loam and clay loam, undifferentiated	Grassland	Soil description	2
Altamont loam and clay loam	Grassland	Soil/climate	165
	Arid grassland	Soil/climate	3
Altamont sandy loam	Grassland	Soil/climate	42
	Arid grassland	Soil/climate	2
Antioch loam and clay loam	Arid grassland	Soil/climate	17
	Grassland	Soil/climate	4
Arnold sandy loam	Arid grassland	Soil/climate	3
Capay and Merced clay, undifferentiated	Alkali sink	Soil description	2
Chino and Foster loam, undifferentiated	Wetland	Secondary sources	43
	Alkali sink	Secondary sources	1
	Desert scrub	Secondary sources	1
Corning and Pleasanton loam, undifferentiated	Grassland	Soil/climate	5
	Arid grassland	Soil/climate	1
Cuyama sandy loam and loam	Grassland	Soil/climate	7
	Arid grassland	Soil/climate	2
Delano loam	Arid grassland	Soil/climate	14
	Grassland	Soil/climate	1
Delano sand and sandy loam	Arid grassland	Soil/climate	21
	Grassland	Soil/climate	6
Diablo adobe soils	Vernal pool grassland	Soil description	30
Dublin adobe soils	Grassland	Soil description	20
Dublin and Yolo loam and clay loam, undifferentiated	Grassland	Soil description	1
Ducor loam	Grassland	Soil description	1
Foster sandy loam	Valley oak	Secondary sources	29
	Alkali sink	Secondary sources	4
Fresno and Merced loam, undifferentiated	Alkali sink	Soil description	8
Fresno clay loam	Alkali sink	Soil description	1
Fresno clay loam, dark phase	Alkali sink	Soil description	4
Fresno clay loam, light phase	Alkali sink	Soil description	14
Fresno fine sandy loam, dark phase	Grassland	Soil description	72
Fresno fine sandy loam, light phase	Alkali sink	Soil description	61
Fresno loam, dark phase	Grassland	Secondary sources	48
	Alkali sink	Secondary sources	1
Fresno loam, light phase	Alkali sink	Soil description	5
Fresno sandy loam, heavy phase	Grassland	Soil description	132
Fresno sandy loam, light phase	Alkali sink	Soil description	40
Hanford and Foster sandy loam, undifferentiated	Desert scrub	Soil description	1
Hanford fine sandy loam	Riparian	Secondary sources	68
	Valley oak	Secondary sources	45

Appendix B (continued). Soil series and vegetation classifications for historical land cover.

Soil series	Vegetation class	Primary classification source	N map units
	Grassland	Secondary sources	1
Hanford loam	Alkali sink	Secondary sources	15
	Riparian	Secondary sources	10
Hanford sand	Riparian	Soil description	67
	Desert scrub	Secondary sources	7
	Arid grassland	Secondary sources	3
Hanford sandy loam	Valley oak	Secondary sources	88
	Grassland	Secondary sources	32
	Desert scrub	Secondary sources	11
	Arid grassland	Secondary sources	4
	Alkali sink	Secondary sources	1
Holland loam	Grassland	Soil description	24
Holland sandy loam	Valley oak	Soil description	34
	Grassland	Secondary sources/climate	6
	Arid grassland	Secondary sources/climate	1
Holland sandy loam, dark phase	Vernal pool grassland	Soil description	1
Honcut loam	Valley oak	Secondary sources	2
	Grassland	Secondary sources	1
Kettleman loam and clay loam	Arid grassland	Soil description	55
Kettleman sandy loam	Arid grassland	Soil description	35
Laguna loam and sandy loam	Arid grassland	Soil description	1
Madera and San Joaquin sandy loam, undifferentiated	Vernal pool grassland	Soil description	22
Madera clay loam and clay	Vernal pool grassland	Soil description	30
Madera loam	Vernal pool grassland	Soil description	78
Madera sandy loam	Grassland	Soil/climate	110
	Arid grassland	Soil/climate	3
Mariposa sandy loam and silt loam	Grassland	Soil description	8
Merced clay loam	Wetland	Soil description	11
Merced loam	Wetland	Soil description	15
Mohave sandy loam	Arid grassland	Soil description	3
Montezuma clay adobe	Grassland	Soil description	23
Muck and Peat	Wetland	Soil description	3
Oakdale sandy loam	Grassland	Soil description	39
Oakley and Fresno sand, undifferentiated	Sand dune	Soil description	121
Oakley and Madera sand, undifferentiated	Sand dune	Soil description	66
Oakley sand	Sand dune	Soil description	27
Olympic adobe soils	Grassland	Soil description	19
Olympic loam	Grassland	Soil description	14
Panoche adobe soils	Arid grassland	Soil description	2
Panoche clay loam	Arid grassland	Soil description	32
Panoche loam	Desert scrub	Secondary sources	52
	Arid grassland	Secondary sources	1
	Grassland	Secondary sources	1
Panoche loam and clay loam	Arid grassland	Soil description	33

Appendix B (continued). Soil series and vegetation classifications for historical land cover.

Soil series	Vegetation class	Primary classification source	N map units
Panoche sandy loam	Desert scrub	Soil description	80
Placentia loam and sandy loam	Grassland	Soil description	3
Pleasanton and Antioch loam and clay loam, undifferentiated	Grassland	Soil/climate	9
	Arid grassland	Soil/climate	3
Pleasanton loam and sandy loam	Arid grassland	Soil description	16
Pond clay loam	Desert scrub	Soil description	8
Pond loam	Desert scrub	Soil description	15
Pond sandy loam	Desert scrub	Soil description	7
Porterville adobe soils	Grassland	Soil description	35
Redding gravelly loam	Vernal pool grassland	Soil description	47
Riverwash and Tailings	Riparian	Soil description	4
Rough broken land	Rock	Soil description	64
Rough stony land	Rock	Soil description	113
Sacramento clay	Water	Secondary sources	3
	Wetland	Secondary sources	1
Sacramento clay loam	Wetland	Soil description	9
San Joaquin and Altamont sandy loam, undifferentiated	Vernal pool grassland	Soil description	6
San Joaquin and Madera sandy loam, undifferentiated	Vernal pool grassland	Soil description	1
San Joaquin clay loam and clay	Vernal pool grassland	Soil description	21
San Joaquin loam	Vernal pool grassland	Soil description	105
San Joaquin sandy loam	Vernal pool grassland	Soil description	105
	Arid grassland	Secondary sources	1
Sierra sandy loam	Grassland	Soil description	6
Stockton adobe soils	Grassland	Soil description	9
Stockton and Fresno soils, undifferentiated	Wetland	Soil description	1
Stockton and Madera soils, undifferentiated	Grassland	Soil description	5
Tulare clay	Water	Secondary sources	1
	Wetland	Secondary sources	1
Tulare clay loam	Wetland	Soil description	3
Tulare loam	Wetland	Soil description	3
Tulare sandy loam and sand	Wetland	Soil description	3
	Alkali sink	Secondary sources	1
Water	Water	Soil description	5
Yolo adobe soils	Grassland	Soil description	17
Yolo clay loam	Grassland	Soil description	26
Yolo loam	Grassland	Soil description	26

Notes

California Red-Legged Frog Response to Pond Restoration

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Abstract.—The California Red-legged Frog (*Rana draytonii*) is federally listed as threatened. Habitat alteration is a significant contributing factor in their decline. Pond restoration and enhancement efforts are tools that can reverse this trend by improving habitat conditions that support recovery goals for the species. We removed excess sediment and emergent vegetation at the Garin Newt Pond Wildlife Area in central California to determine if these actions benefited this species. After sediment removal in 2017, the hydroperiod of the pond improved resulting in California Red-legged Frog egg masses and tadpoles increasing by 99% and 97%, respectively, compared to 2008–2016. We also found significant increases in number of adult and larvae sampled pre-restoration (2008 to 2017) and post-restoration (2018 to 2019). Although only one pond, this site-specific information on California Red-legged Frog response to pond hydroperiod improvements in a central California rangeland may assist recovery efforts designed to preserve and manage habitat for this threatened species.

Key Words.—amphibian breeding; conservation; pond hydroperiod improvements; *Rana draytonii*

The California Red-legged Frog (*Rana draytonii*) was once abundant in central California, ranging from the coast to the Sierra Nevada foothills, but is now considered a threatened species (U.S Fish and Wildlife Service [USFWS] 1996). This frog inhabits permanent and seasonal water sources (streams, lakes, marshes, natural and human-made ponds, and ephemeral drainages) in valley bottoms and foothills up to 1,500 m in elevation (Bulger et al. 2003; Jennings and Hayes 1994). The terrestrial habitat used by this species generally includes an abundance of cover (e.g., burrows, wood debris, and vegetation) in close proximity to water (Fellers and Kleeman 2007; Tatarian 2008; USFWS 2002). Its diet is dependent on prey availability, but mostly consists of terrestrial invertebrates (Bishop et al. 2014).

Breeding adult California Red-legged Frogs (Fig. 1) make use of a variety of aquatic habitats, while larvae use streams, deep pools, and the backwater areas of creeks, ponds, marshes and lagoons (Thomson, et al. 2016). Livestock ponds commonly serve as breeding sites when they provide the proper hydroperiod and pond and vegetative structure (Ford et al. 2013). From November through April, breeding adults may be observed in still or slow-moving water with light to dense riparian or emergent vegetation, such as cattails (*Typha* spp.), tules (*Scirpus* spp.) and willows (*Salix* spp.; Hayes and Jennings 1988). Their egg masses are attached to plants below the surface of the water and hatch after 6–14 d (Jennings and Hayes 1994; Storer 1925). Following hatching, larvae undergo metamorphosis within 3.5–7 mo and reach sexual maturity at 2–3 y of age (Thomson et al. 2016; Jennings and Hayes 1994).

The California Red-legged Frogs is the largest native frog in California but has disappeared from over 70% of its historical range (Fisher and Shaffer 1996; Hayes and Jennings 1986). Major factors that have contributed to this decline include habitat loss, habitat fragmentation, and introduction of the invasive American Bullfrog (*Lithobates catesbeiana*; Hayes and Jennings 1986; Lawler et al.1999). Altered pond sedimentation loading and inundation period are two other leading stressors associated with amphibian declines (Richter et al. 1997).

Restoration efforts aimed at improving the conditions for native species should be judged by how successful wildlife species respond to such attempts (Morrison

Figure 1. California Red-legged Frog (*Rana draytonii*) from the Garin Regional Park, California. (Photographed by Daniel I. Riensche).
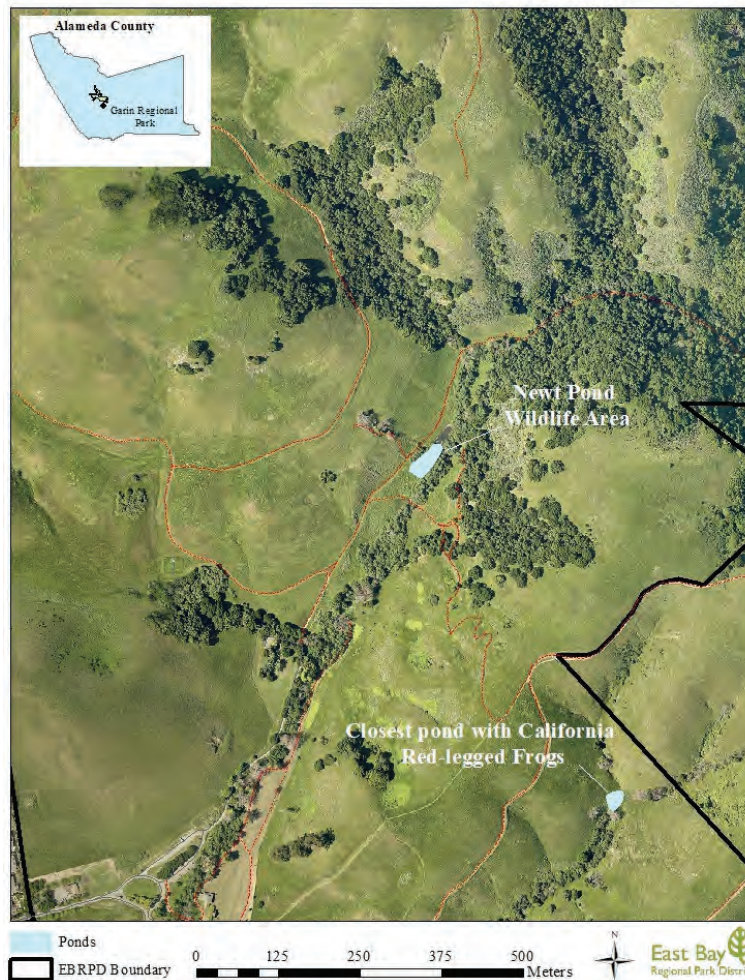
Figure 2. Site map of the Garin Regional Park, Hayward, California, where we conducted a pond restoration project at the Newt Pond Wildlife Area.

2002). The benefits of restoration may increase over time due to system stability and generation succession (Rowe and Garcia 2014). After almost a decade (2008–2016) of us observing only the occasional adult California Red-legged Frog in the seasonal pond known as the Garin Newt Pond Wildlife Area (GNPWA; managed by the East Bay Regional Park District) with no evidence of reproduction except in the 2017 season, we embarked on an effort to improve the breeding habitat and larvae survival of the species. Here we report on how the California Red-legged Frog responds to improved pond hydroperiod resulting from the removal of excess sediment and emergent vegetation.

We studied frogs at the GNPWA (37°38'7.02"N, 122°1'28.44"W), which is part of the Garin Regional Park (a wildlands area) located in Hayward, California (Fig. 2). This lentic environment (0.2 ha) is situated between the transition of Oak Woodland, Grassland, and Riparian habitat types. Earning its name from its importance to California Newts (*Taricha torosa*), Auza (1969) reported that more than 1,600 newts annually traveled 3.2 km (2 mi) or more to breed at the GNPWA. Other wildlife species making use of this seasonal pond area

included: Black-tailed Deer (*Odocoileus hemionus*), Golden Eagles (*Aquila chrysaetos*), Western Fence Lizards (*Sceloporus occidentalis*), Pacific Tree Frogs (*Pseudacris regilla*), and California Red-legged Frogs (Cogswell 1966; Samuel McGinnis, unpubl. report). This site is within the California Red-legged Frog critical habitat unit ALA-1B (USFWS 2010). Due to the seasonal nature of the pond, there are no American Bullfrogs found at this site.

We conducted surveys for all life stages of the California Red-legged Frog from 2008 to 2019, from January through June, using the standardized habitat assessment and protocol-level survey guidelines (USFWS 2005). Daytime visual encounter surveys for egg masses and adults occurred twice monthly (January through June, ≥ eight visits), with no more than 14 d between visits (we did not conduct nighttime surveys). During these assessments, we used binoculars (10 × 42 mm) to search the pond bank, water, floating and emergent vegetation, and woody debris, to sight adult frogs or egg masses.

Due to staffing limitation, from 2008–2019, we conducted one survey annually in the pond for the presence of larval amphibians (in early May). Prior to restoration, the
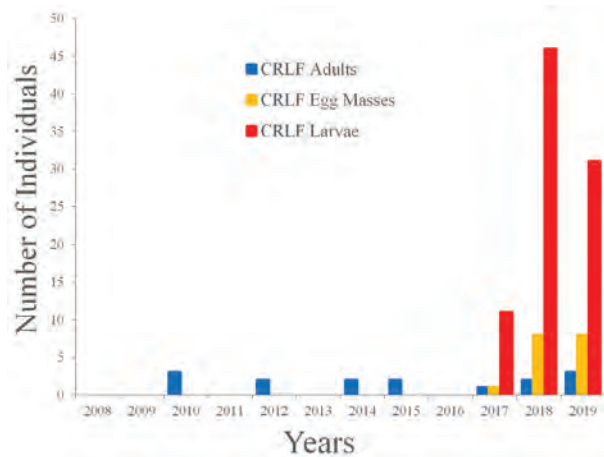
Figure 3. Adults, egg masses, and larvae of California Red-legged Frog (*Rana draytonii*; CRLF) at the Newt Pond Wildlife Area, California, before (2010–2017) and following pond restoration (2018–2019).

GNPWA pond was very shallow (ranging from 0.25–0.5m in depth), choked with Cattails (100% vegetative cover), and did not maintain water throughout the spring (the pond typically dried by mid-to-late June) and supported few adult frogs (Fig. 1). Due to the exceptional rainfall in 2017, when the site received 150% of its normal annual precipitation (https://water.weather.gov/precip/index), this was the only record of larval production in a decade (Fig. 2). In all years, we sampled the pond using D-shaped (radius 23 cm), long-handled dipnets (3.2 mm mesh). Dipnet sweeps consisted of moving a net 1.5 m through representative portions of aquatic features, thus sampling 28 L of larval amphibian habitat per sweep. To minimize disturbance of pond flora and fauna, we stopped sampling after 10 dipnet sweeps if we found California Red-legged Frog tadpoles. We recorded all life stages of frogs, the number of individuals found, and size classes.

The pond restoration efforts that occurred September 2017 included the removal of 199 m3 of sediment and cattails by use of a long-reach excavator, loader, and dump truck. We dredged to improve the hydroperiod of the pond to make it 1–1.5 m in depth (from early April into mid-May) and to remove upwards of 80% of the vegetative cover (thick stand of cattails that covered 100% of the pond surface area). After dredging, the pond held water into late July and early August in 2018 and 2019. We collected population data preceding and following the restoration by using the systematic survey protocols (see above). To compare the reproductive output (average number of egg masses and larvae) before (2008–2017) and after (2018–2019) restoration, we used a two-sample *t*-test with α = 0.05. Since the restoration effort in the fall of 2017, the California Red-legged Frog population at the GNPWA significantly increased in the average number of egg masses (t = -5.73 df = 10, P < 0.001) and the average number of larvae (t = -6.27, df = 10, P < 0.001; Fig. 3).

Excess emergent vegetation is recognized as creating detrimental habitat conditions for the reproductive output of California Red-legged Frogs because it prevents the surface water from reaching suitable temperatures for larvae development (Norman Scott, pers. comm.). Further, sediment removal can create greater seasonal pond depth, thus increasing the inundation period during the spring and early summer, and its potential for successful native amphibian development. American Bullfrogs tend to inhabit ponds changed in some way by humans and they breed in perennial ponds (D'Amore et al. 2010; Doubledee et al. 2003). Apparently, de-sedimentation and the removal of excessive emergent vegetation in seasonal ponds favors the California Red-legged Frog because this species tends to breed and lay its eggs in deep water (Natural Resources Conservation Service 2006; Bradely Shaffer and Robert Fisher, unpubl. report).

To restore a successful breeding population of California Red-legged Frogs at the GNPWA, we removed sediment and cattails to increase the overall depth of the pond and lengthened the amount of time water would be held in this seasonal pond, until early summer. Restoration at this one pond increased the number of egg masses and larvae of California Red-legged Frogs over the two-years following this action, suggesting that regular pond maintenance (sediment and emergent vegetation removal) can be an effective management tool that may benefit this threatened species at similar aquatic sites. Generally, a pond restoration project with the appropriate operations and maintenance has a lifespan of about 20 y (Jackie Charbonneau, pers. comm.).

Such restorative efforts may increase benefits over time because frogs born in a certain pond are likely to remain and have offspring of their own in the same location. Tatarian (2008) reported that most tagged California Red-legged Frogs in her study did not migrate from their source pool over two seasons. Likewise, Feller and Kleeman (2007) reported that only a few of the 123 California Red-legged Frogs studied in Marin County, California, moved farther than the nearest suitable non-breeding habitat. In their study, the furthest distance traveled was 1.4 km and most dispersing frogs moved through grazed pastures to reach the nearest riparian habitat (Feller and Kleeman 2007). Bulger et al. (2003) suggested that breeding sites should take priority in restoration planning because they will allow the species to recover in population size.

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Wildlife Mortalities in Open-Topped Pipes in Central California

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Abstract.—Nearly 20 billion birds are killed in the United States each year by a number of anthropogenic causes, but a lesser known threat is open-topped pipes. Open-topped pipes are prevalent across the landscape, as they are used for a multitude of purposes including agriculture, mining, and infrastructure. Birds, herptiles, and small mammals can be attracted to the pipes as sites for nesting or shelter but they soon can become trapped by the smooth interior and small diameter. Cavity-nesting birds are the most likely animals to enter these pipes, potentially due to competition over a decreasing number of tree cavities in their natural habitat. We inspected pipes in several areas in central California including the South Fork Kern River Valley, Ridgecrest, and Fresno. The main goal of this study was to document the prevalence and predominant taxa comprising the wildlife mortalities in pipes, while also examining the influence of pipe dimensions and adjacent landscape types. We found 13.3% of pipes studied caused mortality: the majority of those mortalities were birds and significantly more deaths were in desert landscapes. The results of this study reveal the need for future research and pipe alteration projects to prevent further mortalities, especially in high priority desert shrubland areas.

Key Words.—birds; cavity-nesting; mortality; desert shrubland; herptiles; mining claim posts; mammals; PVC pipes.

Introduction

It is estimated that up to 20 billion birds are killed in the United States each year due to direct anthropogenic causes, including collisions with vehicles and various manmade structures, poisoning by oil spills and other contaminants, and predation by domestic cats (Loss et al. 2012, 2015). A lesser-known threat to birds are hollow metal or PVC (polyvinyl chloride) pipes or posts, which have the potential to kill a large number of birds annually (Ogden 2013; American Bird Conservancy 2016; Malo et al. 2016). Open-topped pipes, found on farms, ranches, oil production facilities, construction sites, residential areas, and rooftops, serve a variety of purposes including fencing, irrigation, plumbing, ventilation, and mining claim markers. Birds, small mammals, and reptiles enter the pipes to nest or find shelter, but the smooth interior and tight confines of the pipes prevent individuals from escaping, leading to a slow death by stress, dehydration, or starvation (Brattstrom 1995; Hathcock et al. 2014; Malo et al. 2016; Peter Bradley and Jason Williams, unpubl. report). In 2017, the Bureau of Land Management (BLM) estimated that there were 3.6 million mining claims (each with four or more markers) in the western United States, with Nevada having about a third of the claims (1.1 million; Bureau of Land Management [BLM]. 2018. Public Land Statistics 2017. Available from https://www.blm.gov/about/data/public-land-statistics. [Accessed 22 March 2019]). Previously these markers were often made of wood, but in the 1970s they started being replaced by PVC pipes, which were typically hollow and open-topped with about a 10-cm diameter opening (Wilshire et al. 2008). These mining claim markers and other open-topped pipes present a potentially large scale and wide ranging threat to birds in the western United States.

The Nevada Department of Wildlife (NDOW) conducted one of the first studies on bird mortalities in open-topped mining claim markers (Peter Bradley and Jason Williams, unpubl. report). The study began in 1986 after a local chapter of the Sierra Club reported that Mountain Bluebirds (*Sialia currucoides*) were found trapped in mining claim markers. NDOW biologists found 914 dead birds representing 33 species in 7,058 posts in northeast Nevada and estimated that there were 13 dead birds per 100 posts (Peter Bradley and Jason Williams, unpubl. report).

California had an estimated 320,617 mining claims (BLM, *op. cited*) in 2017, which could total to an estimated 1.2 million potential open-topped markers. The issue was first addressed in California in 1990 by LaPre (1990), who reported that 262 dead birds and lizards were found in 820 mining claim posts in the Eastern Mojave National Scenic Area. This study prompted the BLM in California to conduct their own survey of 750 mining claim markers and they found 25% had dead birds and lizards (unpubl. report). Subsequently, California passed legislation that required using solid metal or wooden posts or mounds of stone when marking mines (State of California 1991; Baicich 2012); however, open-topped pipes can still be found in California (e.g., mining claims placed prior to 1991) or pipes that are used for other purposes (irrigation vent pipes, fence posts, rooftop vent pipes).

Across news articles and Audubon newsletters, the prevalence of bird mortalities in open-topped pipes were reported as being quite substantial, although variable. Compared to the news media and gray literature, peer-reviewed scientific research documenting wildlife mortality in open-topped pipes in North America is limited. One of the few studies published was conducted in the mountains of eastern San Bernardino County in California (Brattstrom 1995). In this study,

140 PVC mining claim posts were searched and found to contain the carcasses of birds (19%), lizards (28%), and mammals (4%). Another study, in north central New Mexico on the Los Alamos National Laboratory property, searched open bollard pipes and open pipes on gates and found 19.6% of the 188 pipes had dead birds (Hathcock and Fair 2014). Similar to the study conducted by the NDOW (Peter Bradley and Jason Williams, unpubl. report), both Brattstrom (1995) and Hathcock and Fair (2014) found that a large majority of the dead birds identified in pipes were native cavity-nesting songbirds: Mountain Bluebirds in Nevada, Ash-throated Flycatchers (*Myiarchus cinerascens*) and Cactus Wrens (*Campylorhynchus brunneicapillus*) in California, and Western Bluebirds (*Sialia mexicana*) in New Mexico.

The habitat surrounding open-topped pipes are likely to influence the diversity and abundance of species that are attracted and trapped. Tree cavity shortages limit the numbers of hole-nesting birds an area can support, as several species can compete to use the same sites (Newton 1994). Lower habitat quality and increased competition can drive cavity-nesting birds to occupy manmade holes. In the case of nest boxes, Mänd et al. (2005) found a greater occupancy of nest boxes placed near deciduous habitat where a higher number of cavity-nesting species existed, compared to nest boxes placed near less diverse coniferous habitat. In some cavity nesting species such as bluebirds (*Sialia* spp.), use of artificial cavities is most frequent in areas with perches, wooded pastures, high grass and shrub availability, and sparse ground cover (Munro and Rounds 1985; Hsu and Humpert 1988).

We documented wildlife mortalities in various types of open-topped pipes in several areas in central California: the Kern River Valley, Ridgecrest, and Fresno and King counties. The objective of this study was to quantify the prevalence of wildlife mortalities in open-topped pipes in central California to add to the more extensive work done in Nevada. We examined how several factors may have influenced the prevalence of mortalities in pipes including: (1) taxonomic group (birds, herptiles and mammals); (2) pipe dimensions (height and diameter); and (3) surrounding habitat type (orchard, agriculture, riparian or desert). The results of this study will add to the understanding of the potential threat that open-topped pipes pose to wildlife.

METHODS

Study site. —We conducted this study in four areas in central California (Fig. 1). The first area was in the South Fork Kern River Valley on the lands adjacent to Audubon's Kern River Preserve (35.6690N, 118.3050W), California Department of Fish and Wildlife Canebrake Ecological Reserve, and the South Fork Wildlife Area of the U.S. Forest Service. This area encompasses approximately 20 km of contiguous riparian forest (Fig. 1). Kern River Preserve employees have covered most

of the open-topped pipes on the preserve itself (unpubl. report), but the surrounding grazing pasture upstream of the Kern River Preserve, and the adjoining South Fork Wildlife Area remained undocumented. The South Fork Kern River Valley is 16 km long and 800 m in elevation and is located at the southern end of the Sierra Nevada, and has been designated as an important area for birds (National Audubon Society [NAS]. 2017. Important bird areas: South Fork Kern River Valley. NAS. Available from https://www.audubon.org/important-bird-areas/south-fork-kern-river-valley. [Accessed 26 November 2017]). The riparian habitat in the SFWA is composed of Fremont Cottonwoods (*Populus fremontii*), Red Willows (*Salix laevigata*), and Goodding's Black Willow (*Salix gooddingii*) as the canopy, and Coyote Willow (*Salix exigua*), Mule Fat (*Baccharis salicifolia*), Stinging Nettle (*Urtica dioica holosericea*), Mugwort (*Artemesia douglasiana*), and a variety of grasses and forbs as the understory (Whitfield et al. 1999). The forest is intermixed with freshwater marshes characterized by cattails (*Typha* spp.) and tules (*Scirpus* spp.; Whitfield et al. 1999). The area supports a diverse range of wildlife species that are potentially at risk of entrapment in pipes, including several small rodents (e.g., *Peromyscus* sp.), fence lizards (*Sceloporus* sp.), and over 339 birds (Hewett 1984). A number of cavity-nesting birds exist in the area, such as flycatchers, bluebirds, woodpeckers, swallows, chickadees, wrens, kestrels, and owls.

In addition to the South Fork Kern River Valley, we inspected pipes in nearby areas with similar landscapes and characteristics, including near Ridgecrest (35.9749N, 117.3540W) within Kern County and Sanger (36.7080N, 119.5560W) within Fresno County, and near Riverdale (36.3008N, 119.7829W) within Kings County (Fig. 1). We examined mining claim markers on BLM land north of Ridgecrest and west of the China Lake Naval Air Weapons Station. The area is situated at the edge of the Mojave Desert and the foothills of the Sierra Nevada and is characterized as high desert shrubland with rocky hills. We found pipes near Riverdale that were situated in agricultural fields and in small, neighborhood orchards and vineyards in Sanger. These two cities lie within the San Joaquin Valley and have relatively flat landscapes. Their semi-arid climates include hot, dry summers and mild, rainy winters (Tucker 2013). A number of cavity nesting birds, as well as small mammals, reptiles, and amphibians inhabit these areas.

Data collection. —We collected data from the middle of May through August 2017. We conducted weekly roadside surveys in search of pipes. We planned the general survey locations in advance to contact any known landowners for permission to access their properties. When a pipe was spotted, we pulled over and initially documented what the pipe was being used for, assigned it a number, and used a GPS unit (Garmin GPSMAP 76CSx, Garmin Ltd., Olathe, Kansas) to mark location
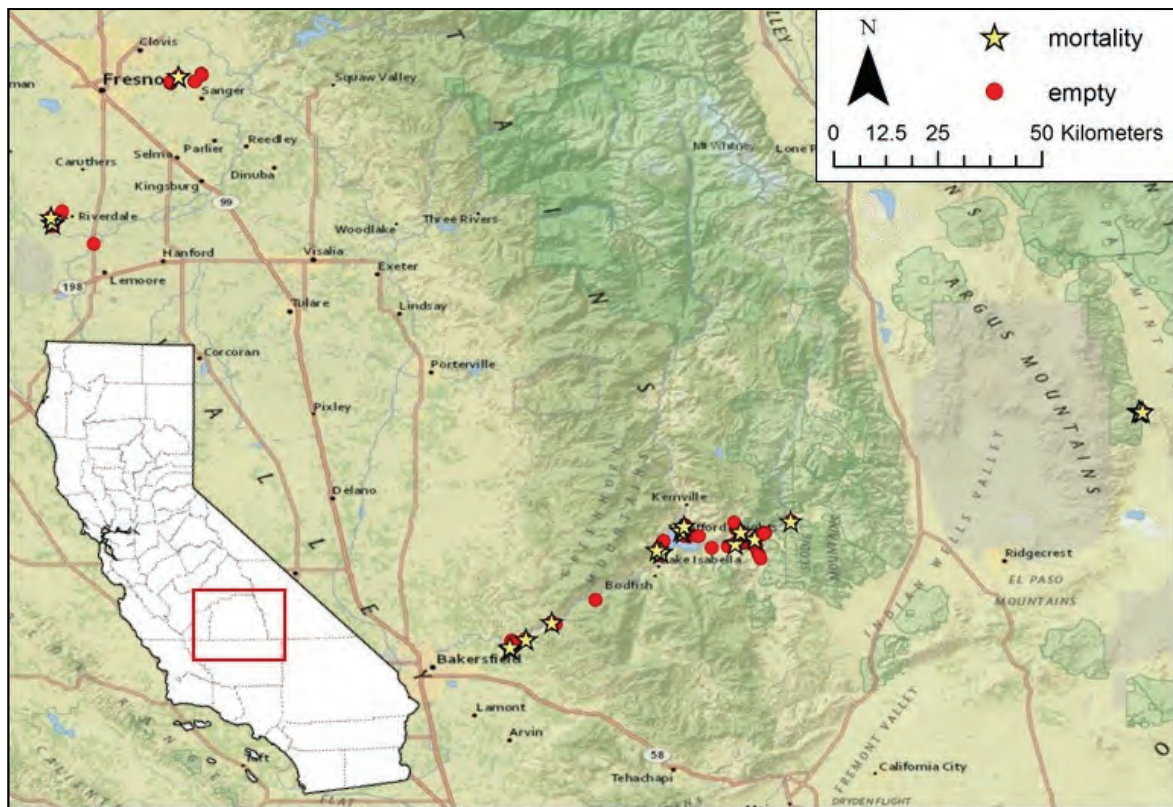
Figure 1. The locations of the pipes checked May-August 2017 in the Kern River Valley and Ridgecrest in Kern County, Sanger in Fresno County, and near Riverdale in Kings County, California, representing those empty and those with wildlife mortalities.

coordinates. We checked an average of 41 pipes during a full day of search effort. Most pipes were gate markers or fence posts, but we also surveyed irrigation pipes, livestock corrals, signposts, BLM mining claim markers, and lone pipes that appeared to serve no function. Of the 295 pipes inspected, 256 (87%) pipes were made of metal and only 39 (13%) were of plastic material.

We documented several factors about each pipe and its location in order to assess how they might affect wildlife mortalities. The diameter and length of each pipe were measured with a measuring tape (length was taken from the ground to the top of the pipe that was above ground and thus did not include the portion below ground) and classified position as upright, horizontal, or leaning. The majority of the pipes were positioned upright (277) with only 12 leaning and six horizontal; therefore, we did not include this characteristic in our analyses. We also described the habitat and vegetation components surrounding each pipe based on general visual observations, noting whether the pipe was near roads, buildings, cow pastures, forests, or orchards and then used this information to categorize each pipe into a habitat type category.

The contents of each pipe were searched for any signs of wildlife, dead or alive, using a 700+ lumen flashlight. The flashlight allowed us to identify the presence or absence of an animal, as well as categorize what taxa it belonged to and the number of individuals inside of a single pipe. The presence of trash and debris was also noted, as this could potentially affect our ability to

accurately estimate taxa or number of individuals.

Data analysis.—Because we did not necessarily predict a linear relationship of pipe diameter and height with wildlife mortality, we grouped the data by pipe diameter and height into interval categories depending on the range of values obtained. We expected that the pipe frequencies would not be equal across categories, but we tried to make it so that each category contained an adequate number of pipes so a pattern of wildlife mortality could be detected during analyses. We ended up using three diameter intervals of 2.5–8.33 cm (small), 8.34–14.16 cm (intermediate), and ≥ 14.17 cm (large), and the numbers of pipes that fell within each category were fairly equal (Table 1). The height variable, however, was more uniform (most pipes were 120–160 cm tall), therefore we only split height into two categories, 12.7–120.9 cm (short) and 121.0 cm and taller (tall); and there were far fewer pipes in the short height category than the tall category (Table 2).

In addition, we grouped the habitat descriptions into landcover categories using available types within the study areas. These categories included: urban, agriculture, orchard, riparian/temperate forest, and desert shrubland; however, only two pipes could be classified as urban, so that category was removed from the analysis. We defined agriculture as a crop field, cow pasture, or otherwise open, grassy area. An orchard contained uniform rows of trees or vineyards. We defined riparian/temperate forest by groupings of trees > 5 m tall, an understory of shrubs or saplings, and possible water bodies. Desert shrubland

TABLE 1. The number of pipes in each diameter category (cm) comparing those empty and those with wildlife mortalities measured May-August 2017 in the Kern River Valley and Ridgecrest in Kern County, Sanger in Fresno County, and near Riverdale in Kings County, California.

Diameter (cm)	Mortality		% Yes	Total
	No	Yes		
Small (2.50–8.33)	75	5	6.67	80
Intermediate (8.34–14.16)	97	22	22.60	119
Large (≥ 14.17)	84	12	14.20	96
Total	256	39	13.22	295

was an open, rocky area with sparse vegetation in the form of short shrubs. These habitat types followed a gradient of human disturbance, with agriculture and orchard sites considered highly impacted by humans, while forest and desert sites were considered to have less human influence. We placed pipes located in areas with characteristics that fell into multiple categories in the habitat type that was most dominant and influential. For example, we classified a roadside pipe near both a pasture and a riparian forest in the riparian/temperate forest category, as the presence of trees was deemed more influential on the types of species that might use the area (e.g., cavity nesting bird species). The numbers of pipes inspected in each habitat category were fairly evenly distributed with 76 pipes in orchards, 73 in agriculture, 59 in riparian/forests, and 87 in desert areas.

We used a series of Chi-square tests to compare the frequency of the response variable wildlife mortality in a pipe (yes or no) across our three categorical independent variables including pipe diameter, pipe height, and habitat type. We also calculated the effect sizes using Cramer's V for each analysis which allowed us to determine the strength of association of any significant results (Cohen 1988). We used SPSS (IBM Corporation) for all statistical analyses with $\alpha = 0.05$.

RESULTS

We inspected 339 pipes, 295 of which we were able to determine if there were wildlife mortalities. We excluded the additional 44 pipes from the analysis because it could not be determined with confidence whether the pipes contained wildlife. This was due to trash and/or debris obstructing the view inside the pipes.

Of the 295 pipes we inspected, 39 (13.3%) contained dead wildlife. Twenty-one pipes (7.1%) contained signs of birds, specifically passerines, including full carcasses, feathers, and a nest with a cracked egg (Appendix Fig. 1). Although most birds could not be identified to species, two feathers were confirmed as belonging to a Western Bluebird and a Western Meadowlark (*Sturnella neglecta*). We detected herptiles in five pipes (1.7%), including five Western Fence Lizards (*Sceloporus occidentalis*), a Western Toad (*Anaxyrus boreas*), and a Great Basin Gopher Snake (*Pituophis catenifer deserticola*; Appendix Fig. 2). We found mammals in nine pipes (3.1%), all of which were mice (*Peromyscus* spp.; Appendix Fig. 3). Four pipes (1.4%) held unknown carcasses that we could not identify beyond vertebrate status, including a spine, skulls, and other assorted bones.

Pipe diameter category significantly influenced the frequency of wildlife mortality in pipes ($X^2 = 6.309$, df = 2, $P = 0.043$) but the strength of this association was weak (Cramer's V = 0.146; Cohen 1988). There were more deaths in pipes with an intermediate-sized diameter (8.34–14.16 cm) than the small or large pipes (Table 1). There was no significant difference in the number of pipes with wildlife mortality of differing heights ($X^2 = 1.338$, df = 1, $P = 0.223$; Table 2). Habitat type had a significant influence on the number of pipes with wildlife mortality ($X^2 = 10.598$, df = 3, $P = 0.014$) and the effect was moderate (Cramer's V = 0.190; Cohen 1988). Specifically, pipes in desert landscapes had more wildlife deaths than those in orchard, agricultural, or forested areas (Fig. 2).

DISCUSSION

We found that just over a tenth of pipes inspected in our central California study area had wildlife mortalities, with birds being the greatest represented taxonomic group (7.1%), followed by mammals (3.1%) and herptiles (1.7%). These results are lower than previous studies conducted in California in San Bernardino County (LaPre 1990; Brattstrom 1995), in particular the number of pipes with herptiles (e.g., Brattstrom 1995 found 28% of pipes inspected had

TABLE 2. The number of pipes in each height category (cm) comparing those empty and those with wildlife mortalities measured May-August 2017 in the Kern River Valley and Ridgecrest in Kern County, Sanger in Fresno County, and near Riverdale in Kings County, California.

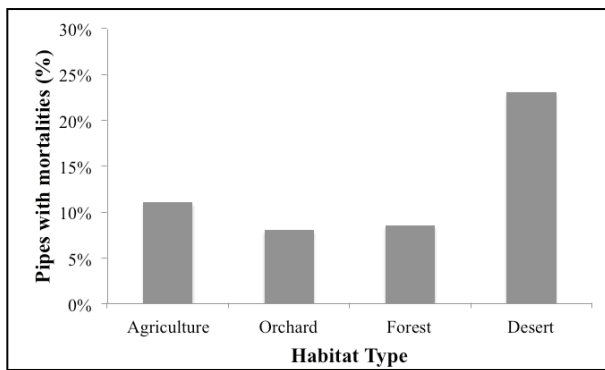| Height (cm)	Mortality		% Yes	Total
	No	Yes		
Short (12.70–120.9)	82	9	9.89	91
Tall (≥ 121.0)	174	30	14.70	204
Total	256	39	13.22	295

Figure 2. The percentage of pipes with wildlife mortalities in four habitat types observed May-August 2017 in the Kern River Valley and Ridgecrest in Kern County, Sanger in Fresno County, and near Riverdale in Kings County, California. The total number of pipes inspected in each habitat were 73 in agriculture, 76 in orchard, 59 in forest/riparian, and 87 in desert.

lizards). This difference may be due to differences in habitat types as well as distribution and abundance of species in Kern, Kings, and Fresno counties compared to San Bernardino.

We found differences in habitat types in our study with the largest proportion of wildlife deaths in the pipes found in the desert shrubland landscape type, which may in part be explained by the observed scarce vegetation and therefore potential lack of natural cavities in those areas. Orchards and riparian/temperate forests provide trees for nesting and roosting that may be preferred over man-made holes. Agricultural areas are similarly open landscapes with a lack of trees and natural cavities or shelter, but desert shrubland represents a more natural habitat type and may have more native species. In contrast, the Nevada Department of Wildlife found a positive relationship between the distances to Pinon-Juniper Woodlands with bird mortality in open-topped pipes; however, mortalities were also found in Sagebrush Steppe and Salt Desert Shrub landcover (Peter Bradley and Jason Williams, unpubl. report). It is likely that across species these relationships will vary as some species will prefer to use natural areas, whereas others may also use disturbed areas and perhaps even expand into disturbed areas because of human structures (e.g., Dunning and Bowers 1990) and in some cases can benefit from new nesting structures (Morelli et al. 2014). We did not systematically survey the presence of natural cavities across our sites in this study, and thus cannot quantify what options were available for cavity nesting birds; however, future studies should consider this variable to test if entrapment in pipes is greater in areas with fewer natural cavities.

The structure, dimensions, and type of some pipes may be more hazardous for certain wildlife species than others. Overall, we found that intermediate-sized diameter (8.34–14.16 cm) pipes were more likely to have wildlife mortalities. This may simply be due to easier access; however, it was more difficult to clearly see to the bottom of pipes with smaller diameters so this may have

also played a role in this relationship. For mammals, we found that four of the nine pipes we inspected that caused mice mortalities were irrigation pipes dispersed throughout a Sequoia National Forest campsite. We found the remains of several mice in each individual pipe, along with a western toad and a bird. We did not find a distinct pattern for herptiles, but Brattstrom (1995) found all pipes with lizards were perforated posts. Although we only examined very few horizontal pipes, one could predict that wildlife may have an easier time escaping compared to vertical pipes. Nevertheless, Brattstrom (1995) discovered a dead Desert Cottontail (*Sylvilagus audubonii*) in a post laying on the ground, and an Audubon employee found over 200 birds in a fallen irrigation standpipe on the Kern River Preserve (Audubon, unpubl. report), but in the latter case the pipe was previously upright.

The areas with the most mortalities for wildlife in our study was located on BLM land near Ridgecrest, where six out of 11 pipes (54.5%) contained at least one bird. These pipes were previously used as mining claim markers and were dispersed along the crest of a rocky hill far from any urban structures. A BLM employee estimated that at least half of the several dozen mining claim markers he had capped in the surrounding area contained dead birds (Robert Enriquez, pers. comm.). Nationwide, BLM registered 3.6 million mining claims in 2017 (BLM 2018, *op. cited*), which could represent millions of death traps for wildlife. Moreover, mining claim markers are only a small subset of the open-topped pipes used for a variety of other purposes. Comparing the extent of mortality across pipes of different uses and structures, such as mining claim markers, gate markers, vent pipes on buildings and irrigation pipes, would be interesting for a future study.

This study was limited to presence or absence of wildlife mortalities in pipes because pipes were inspected with a flashlight and contents were not removed. We were only able to count the actual number of individuals within a few pipes, so it is unknown if pipes contained one or more individuals. The data likely underestimate the number of pipes with wildlife mortalities, as well. Even though pipes obscured by trash and debris were removed from the analysis, we could have easily overlooked carcasses in seemingly empty pipes. Hathcock and Fair (2014) also used a flashlight to inspect bollards and gate markers except for the few that were removable. Their findings were fairly similar, with positive detections in 11% of gate markers and 27% of bollards. Malo et al. (2016) were much more thorough when investigating uncapped tubular poles along the Madrid-Levante high-speed railway line in central Spain. They examined poles using flashlights and a borescope and extracted carcasses with wire hooks that were later identified. Their findings were significantly higher, with one or more bird remains found in 70 out of 96 poles (72.9%) for a total of 162 carcasses.

Solutions for decreasing open-topped pipe mortalities include filling, crimping, capping, or removing unused pipes, which for mining claim markers in California and Nevada is now required by law (State of California 1991; State of Nevada 2009). In the case of PVC pipes used for mining claims, capping pipes with plastic caps has been shown to not be effective as they are often not monitored once installed and often fall off due to desert weather wear (Peter Bradley and Jason Williams, unpubl. report). Removal of unused pipes or replacing them with other mine markers (wooden posts or rock piles) is obviously the most effective permanent solution. Taking into account labor and material costs, capping existing poles in the field is much more costly than sealing them in the factory in the first place (Malo et al. 2016) or using alternatives that are not open-topped. Solutions for other types of open-topped pipes vary and include securing metal caps for chain link fence posts, filling pipes that cannot be removed with sand or concrete, and covering rooftop and heating vents with galvanized hardware cloth held in place by stainless steel pipe clamps or gutter guard leaf filters (Southern Sierra Research Station. Avian Mortality Epidemic - Death Pipes. Available from http://www.southernsierraresearch.org/Information/DeathPipes/ [Accessed 26 November 2017]). Nevertheless, pipes may still need to be monitored, as installments such as hardware cloth can fall off. One specialized solution that appears very effective has been developed to reduce the entrapment of raptor and other bird species in vault toilets (e.g., pit toilets; Teton Raptor Center. Poo-poo Project. https://tetonraptorcenter.org/our-work/poo-poo-project/ [Accessed 13 December 2017]). Raptors enter the vault toilets through ventilation pipes and the Teton Raptor Center created a stainless-steel screen that secures to the top of the pipe, preventing entry by birds while allowing for ventilation. With the increasing awareness of the issue and solutions being implemented, future research will be able to test the long-term effectiveness of these variety of solutions.

Our study provides insight into the severity of bird and wildlife mortality caused by open-topped pipes in central California. The extent to which these pipes are having a population level effect is unknown (Loss et al. 2015), but the presence of open-topped pipes in areas where threatened or endangered wildlife exists could be of conservation concern. This issue can be invisible to the general public, as wildlife trapped in pipes die completely unnoticed in these hidden locations compared to birds colliding with windows or dead animals brought home by cats. Raising awareness is vital, and a larger dataset of mortalities could attract funding for projects to remove unused pipes and securely close off others. Quantifying the number of potentially threatening pipes that exist would also shed light on the scope of this issue and garner support. Furthermore, open-topped pipes are not the only source of entrapment, as several other human-made structures endanger wildlife such as uncovered trenches dug into the ground (Germano et al. 1993, Germano 1995, Simpson et al. 2011) and oil pits (U.S. Fish and Wildlife Service 2013). Collaboration across federal, state and local natural resource agencies, agricultural workers, landowners, and the public is necessary for successful solutions and preventative measures to be implemented to lessen the impacts of wildlife entrapment and mortality.

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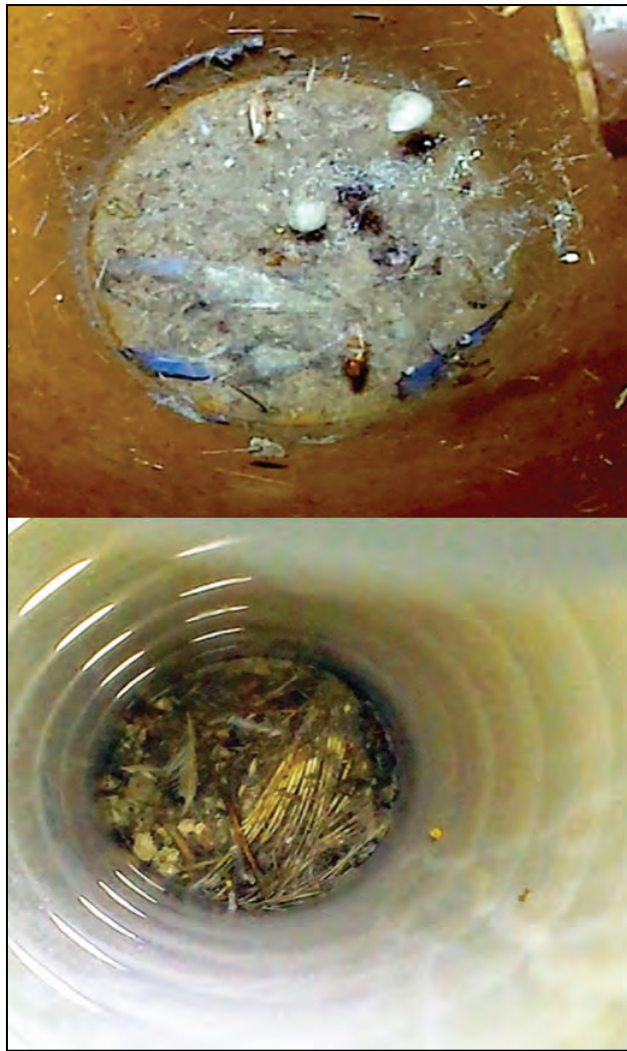
Appendix Figure 1. The contents of a gate marker with (Top) a Western Bluebird (*Sialia mexicana*) and (Bottom) a mining claim marker with an unidentified bird. (Photographed by Michelle Harris).

APPENDIX FIGURE 2. The contents of three metal livestock corral pipes revealing (Top and Bottom Left) two living Western Fence Lizards (*Sceleporus occidentalis*) and (Bottom Right) a Great Basin Gopher Snake (*Pituophis catenifer deserticola*). (Photographed by Michelle Harris).

Appendix Figure 3. The contents of an irrigation pipe revealing several deer mice (*Peromyscus* sp.). (Photographed by Michelle Harris).

Habitat Selection by Newly Metamorphosed Great Basin Spadefoots (*Spea intermontana*): a Microcosm Study

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Abstract.—Wetlands and their surrounding upland areas provide crucial habitat for the Great Basin Spadefoot (*Spea intermontana*), a species at risk in the grasslands of the southern interior of British Columbia, Canada. Understanding habitat selection and how these animals respond to change is crucial in the creation and implementation of conservation plans for the Great Basin Spadefoot. Little is known about the habitat selection of the Great Basin Spadefoot as they transition from aquatic larvae to terrestrial animals. The objective of this study was to determine microhabitat preferences of newly metamorphosed Great Basin Spadefoots. The small size of the metamorphs (18.7 mm SVL ± 2.8 mm (SD; n = 130) and 0.66 g ± 0.33 g (n = 130) precluded telemetry, so we conducted 13 simple habitat selection trials within four artificial enclosures. The trials were run in four batches over a 36-h period in the field near a spawning site. At release in the enclosures, the metamorphs immediately dispersed into all four habitat types (e.g., dry-bare, dry-cover, moist-bare, moist-cover). After 12 h, 75% of the metamorphs were located in moist-cover habitat, indicating a preference for that habitat. At the 36-h time period, it was clear that metamorphs were selecting for moist-cover habitat (79%, *P* < 0.001). Favourable habitat attributes for metamorphic spadefoots may be more specific than that of adults, requiring a more complex, holistic approach to habitat management for the Great Basin Spadefoot.

Key Words.—amphibian; British Columbia; conservation; cover; desiccation; grasslands; juvenile; larval emergence.

Introduction

Habitat selection by juvenile animals may differ substantially from that of adults, particularly for species where parental care is nonexistent or only occurs for very short periods of time (McHugh et al. 2011; Pereira et al. 2019; Delancey and Islam 2019). In situations where there is no parental care, juvenile animals should demonstrate habitat associations and preferences critical to immediate survival and, ultimately, successful recruitment into the population (Patrick et al. 2008). Amphibians provide striking examples of habitat associations: individuals transforming from aquatic larvae to terrestrial animals will abruptly face a different suite of environmental conditions, and the ability to find appropriate habitat may be imperative to survival (Rittenhouse et al. 2008). Understanding habitat selection during this stage of development is important to designing and implementing conservation strategies that do not focus solely on adult habitat needs (Biek et al. 2002; Vonesh and De la Cruz 2002).

North American spadefoots (genera *Scaphiopus* and *Spea*) are terrestrial, burrowing amphibians associated with arid or semi-arid habitats. North American spadefoots rely on two distinct habitats to survive: water bodies for breeding and tadpole development and terrestrial habitat for feeding, aestivation, and hibernation (Pearson 1955). Habitat studies on North American spadefoots have focused largely on adult breeding sites (e.g., Nystrom et al. 2002; Morey and Reznick 2004). As a result,

very little is known about the habitat preference of newly metamorphosed spadefoots (metamorphs), owing in part to their small size (18.7 mm SVL ± 2.8 SD, n = 130, and 0.66 g ± 0.33, n = 130; our data) and cryptic nature. As post-metamorphic juveniles do not remain in the water, they are likely vulnerable to desiccation and predation upon emergence, so habitat available near the edge of water likely plays a key role in enabling some proportion of the animals to survive the critical early stages of terrestrial life (Rothermel and Semlitsch 2006; Roznik and Johnson 2009). Heinen (1993) and Baughman and Todd (2007) conducted lab experiments on recently metamorphosed anurans (American Toad, *Bufo americanus*, Southern Toad, *Bufo terrestris*, and Eastern Spadefoot, *Scaphiopus holbrookii*) and concluded that vegetative cover was chosen over bare ground, and that cover was significant in providing protection from predators; however, neither of these studies included soil moisture as a factor. Jansen et al. (2001) concluded that Eastern Spadefoot metamorphs preferred moist substrate over dry. Grover (1998) analyzed cover and soil moisture and found both to be significant in predicting abundance of both juvenile and adult terrestrial salamanders (Redbacked Salamander, *Plethodon cinereus*, and Northern Slimy Salamander, *Plethodon glutinosus*) in a Virginia forest. Further studies are clearly required to explore the relationship between cover and moisture in determining habitat selection during the critical emergence stage of amphibians. Such work can guide the creation of artificial breeding ponds or the retention of key microhabitats

FIGURE 1. Location of the study site for microhabitat selection of metamorphic Great Basin Spadefoots (*Spea intermontana*) in the semi-arid grasslands in the Thompson River Valley, approximately 10 km west of Kamloops, British Columbia, Canada. A natural spadefoot breeding pond is located in the center of the photo. The four microcosms (arenas) are located to the right of the breeding pond, within the fenced enclosure. (Photographed by Jo-Anne Hales).

during habitat restoration or other land management.

The Great Basin Spadefoot (*Spea intermontana*) ranges further north than any other spadefoot in North America, occupying the semi-arid grasslands of British Columbia (Committee on the Status of Endangered Wildlife in Canada [COSEWIC] 2007; BCSIRAWG 2017). This ecosystem makes up < 1% of British Columbia and has been affected over many decades by various forms of both anthropogenic and natural disturbance, substantially altering the habitat (Wikeem and Wikeem 2004). This includes a dramatic decrease (63%) in grassland ephemeral ponds (Coelho 2015) used by Great Basin Spadefoots. Cattle grazing and water use also has degraded the habitat around many of the remaining ponds (Jones et al. 2011; Teuber et al. 2013). Previous research on the Great Basin Spadefoot in British Columbia is limited and primarily restricted to the movement and breeding ecology of adults (Garner 2012; Ashpole et al. 2014).

Given the small size of metamorphic spadefoots, we hypothesized that newly metamorphosed animals would select habitat that provided cover in the warm, dry climate in the semi-arid grasslands of British Columbia. In this study, we report on a microcosm experiment used to investigate microhabitat preferences of newly metamorphosed Great Basin Spadefoots. The ultimate purpose of this work was to provide information that would help shape emerging guidelines for the rehabilitation, protection, and creation of Great Basin Spadefoot habitat on the British Columbia interior landscape.

METHODS

Study site.—This study took place throughout August 2014 in the semi-arid grasslands in the Thompson River Valley, approximately 10 km west of Kamloops, British Columbia, Canada (50°41'38"N, 120°32'7"W). Typically, summers in the Kamloops area are warm and dry (Chilton 1981). In 2014, Kamloops average daily temperatures in June, July, and August ranged from 11.2° C to 25.8° C, 14.6° C to 32.2° C, and 15.1° C to 29.7° C, respectively (http://climate.weather.gc.ca/index_e.html#access). Total precipitation in June, July, and August was 31.4 mm, 30.5mm, and 51.1 mm, respectively (http://climate.weather.gc.ca/index_e.html#access). Ephemeral alkaline ponds and wetlands are sporadic on the landscape, a feature typical of this zone (Meidinger and Pojar 1991). Due to current and historical cattle grazing, Big Sagebrush (*Artemisia tridentata*), Cheatgrass (*Bromus tectorum*), and Knapweed (*Centaurea sp.)* dominate upland sites. The elevation of the study site is about 643 m. We conducted this work outdoors, in close proximity (< 10 m) to a known breeding pond of Great Basin Spadefoots (Fig. 1) where tadpole development naturally occurred. This outdoor location was open to fluctuating ambient temperature and humidity, and the diurnal cycle.

Microcosm (arena) construction.—We established four enclosures (i.e., microcosms) using plastic arenas
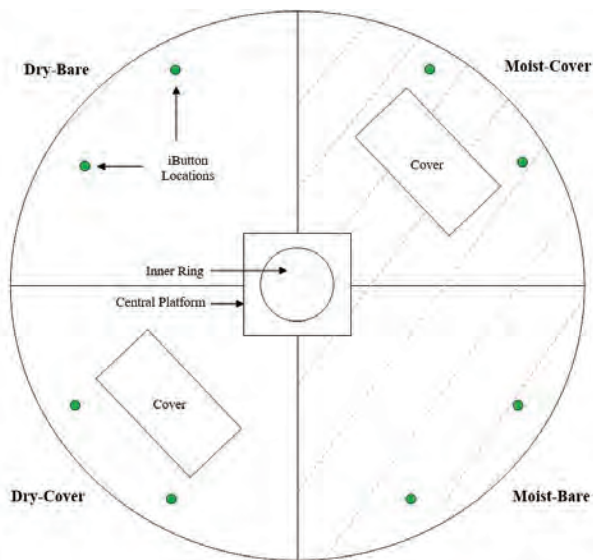
Figure 2. Schematic of artificial enclosure (i.e., microcosm) divided into four habitat types (e.g., dry-bare, dry-cover, moist-bare, moist-cover) to investigate microhabitat selection by newly metamorphosed Great Basin Spadefoots (*Spea intermontana*) in the Thompson River Valley, British Columbia, Canada. Enclosure diameter was 1.2 m. Dashed grey lines define the half of the enclosure that was moistened with water.

(children wading pools) approximately 1.2 m diameter with a wall height of approximate 20 cm. We divided each microcosm into four quadrants (surface area of one quadrant = 0.6 m2) with a central platform of rigid, translucent plastic (about 27 × 27 cm) and inner ring of the same material (about 23 × 23 cm; Fig. 2, 3). We placed the central platform and inner ring at the center of the arena and prevented the metamorphs from burrowing into the soil and entering the quadrants prior to release. Across these quadrants, we established four microhabitat types: dry-bare, dry-cover, moist-bare, and moist-cover. We filled each arena to a depth of 5 cm (volume = 0.06 m3) with a 1:1 mixture of sand and silt obtained from a local gold mining operation (about 4 km south of study site), which was stockpiling the materials for tailings dam construction. We applied 2 L of deionized water to the half of the microcosm containing moist habitat. After water application, soil saturation was visibly evident (Fig. 3). Subsequent applications of water occurred at the 12 h observation to maintain moisture in the soil. We used plastic dividers to prevent moisture from seeping into the dry treatments, but we took care to ensure that there were no physical surface barriers for the metamorphs between the moist and dry halves. Following Baughman and Todd (2007), we used a fresh soil mixture in each trial. The soil mixture was sifted prior to being placed into the arena to ensure all large particles were removed. Light raking ensured an even surface prior to commencement of the trial. We placed pine cover boards (about 15 × 15 × 1 cm) in two of the four quadrants to create covered habitats (pine naturally occurs at the study site and we acquired cover boards locally). We raised

the boards about 2.5 cm above the soil with wooden sticks and positioned stones on top of the cover boards to ensure stability. During the trials, we covered each arena with a metal screen (about 1.5 × 0.8 cm mesh) fine enough to prevent predators from entering the pools and to prevent metamorphs from escaping, but coarse enough to prevent shading.

We sequentially rotated each arena 0°, 90°, 180°, and 270° at the start of each trial to avoid bias in directional orientation. Following Heinen (1993) and Baughman and Todd (2007), the rotation of the arenas also controlled for other possible stimuli in the surrounding environment (e.g., presence of a wetland, direction of the sun). To collect comparative soil temperatures, we buried two Maxim Integrated iButton Thermochron ® temperature data loggers (Model DS1921G, San Jose, California, USA) 2.5 cm deep along the outside edge of each quadrant (Fig. 2, 3). These loggers recorded soil temperatures every 1 h over the 36-h time period. Data from the loggers allowed us to determine whether soil temperatures across the four treatment quadrants differed significantly and therefore possibly exerted an effect on the metamorphs.

Metamorph introduction.—The average size of the metamorphs used in this study at emergence from the water was 18.7 mm SVL ± 2.8 SD (n = 130) and 0.66 g ± 0.33 (n = 130). We fenced two natal ponds (about 2.5 km and about 900 m from microcosms) when tadpoles neared the later stages of metamorphosis to capture the spadefoots immediately upon emergence from the water. We hand captured metamorphs in the evening (about 2000) inside these fences less than 45 min before each series of trials was about to begin. We only selected individuals that had completely absorbed their tail into their body (i.e., Gosner stage 46; Gosner 1960) and that we observed to be physically able to move onto land. We transported animals to the arenas in small plastic containers. Once at the arenas, we weighed individuals (g) on a portable electronic scale (Model 1250BKEF, Salter Housewares Canada Inc., Burlington, Ontario, Canada), measured their snout-vent length (SVL), and tagged them on the right, hind foot using manual injection fluorescent yellow Visible Implant Elastomer (VIE; Northwest Marine Technology, Inc., Shaw Island, Washington, USA). We used the VIE marker to ensure individuals participated in only one trial and were not later recaptured from the natal pond and retested. Each trial consisted of 10 newly metamorphosed spadefoots being introduced into the center of each of the four microcosm arenas. Metamorphs were removed from the arenas and immediately released back into their original natal pond once each trial was complete.

Microhabitat analysis.—We conducted 13 trials (i.e., one set of 10 spadefoots in an enclosure = one trial), run in four batches (i.e., four arenas used simultaneously).

Figure 3. Photographs of artificial enclosures (i.e., microcosms) divided into four habitat types (e.g., dry-bare, dry-cover, moist-bare, moist-cover) to investigate microhabitat selection by newly metamorphosed Great Basin Spadefoots (*Spea intermontana*) in the Thompson River Valley, British Columbia, Canada. A natural spadefoot breeding pond is located within the fenced enclosure, adjacent to the microcosms. (Photographed by Janessa Ekman, left, and Jo-Anne Hales, right).

Each batch took place over two nights, spanning 36 h from start to finish. Prior to initial release into the arena, as per Baughman and Todd (2007), we placed the metamorphs on the central platform and covered them within the inner ring for 10 min, allowing time for the individuals to partially adjust to their surroundings. We then removed the inner ring and cover and visually monitored metamorphs until each had left the central platform and entered one of the four habitat quadrants, which was typically an immediate response. We recorded the location of each metamorph, according to quadrant association, at three time periods: At release, 12 h, and 36 h. At release was defined as the exact moment each individual left the inner ring and entered a quadrant. Our rationale for the later time periods was to provide ample time (including two night periods) for the metamorphs to explore, adjust, and choose a position within their new surroundings.

To determine locations of the metamorphs in the arenas at 12 h and 36 h, we removed the screen gently to prevent disturbance and we completed a visual inspection. When necessary, we briefly removed cover boards and counted the metamorphs. Once we accounted for all 10 individuals in each arena, we gently repositioned cover boards and screens. Extreme care was taken to acquire individual locations as quickly as possible with no disturbance to either the metamorph or the arena. We gently unearthed metamorphs if no sighting could be made but evidence of a burrow existed; we promptly recovered these metamorphs with soil.

Statistical analysis.—We performed statistical analysis using R 3.2.5 (R Core Team, 2015). Prior to conducting analysis, the categorical variables were appropriately designated using the ordered (for ordered categorical [ordinal] variables, e.g., trial number, time period) and factor (for categorical [nominal]) variables, e.g., direction, habitat-type) functions (Kabacoff 2011; Bolker, B.M., M.E. Brooks, C.J. Clark, S.W. Geange, J.R. Poulsen, M.H.H. Stevens, and J-S.S. White. 2011. GLMMs in action: gene-by-environment interaction in total fruit production of wild populations of *Arabidopsis thaliana*, revised version, part 1. Available from www.cell.com/cms/attachment/601623/4742452/mmc1.pdf [Accessed 1 December 2015]). We tested for correlation among our predictor variables (i.e., direction, cover type, soil temperature) by calculating a Generalized Variance Inflation Factor (GVIF) following Zuur et al. (2012) and Fox and Monette (1992). The GVIF measures how much the variance of a predictor is increased due to linear dependence with other predictors. As recommended by Zuur et al. (2010), we used a GVIF value of three as the pre-selected threshold (indicating no collinearity), versus the higher (and more flexible) value of ten used by Montgomery and Peck (1992).

We used Fisher's Exact Tests using the lattice package in R (Sarkar 2008) to explore if metamorphs exhibited equal preference for the four habitat types and directionality in each arena. The results indicated whether data within each batch of trials could be pooled for a robust analysis of metamorph microhabitat preference using
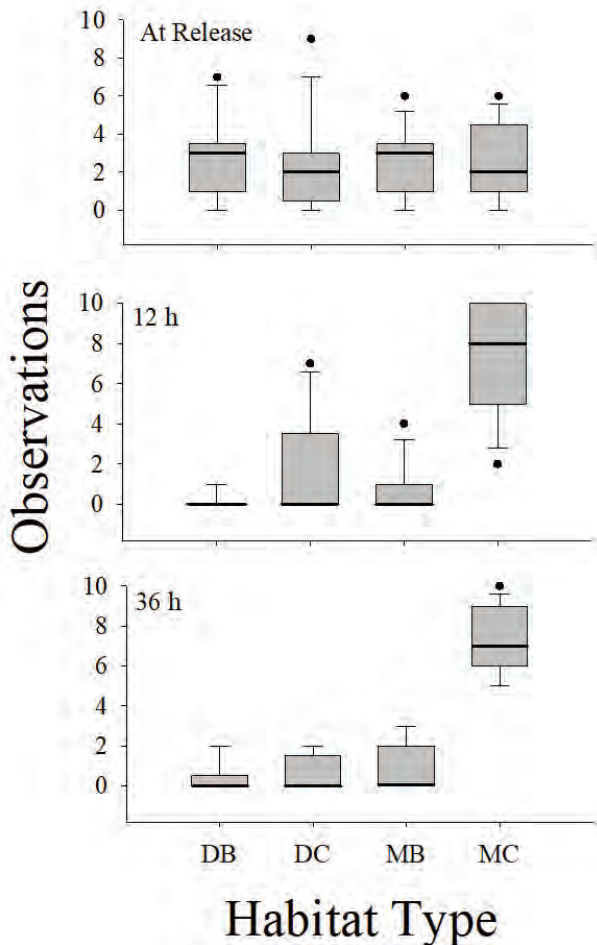
FIGURE 4. Observations of the number of metamorphic Great Basin Spadefoots (*Spea intermontana*; n = 130) in the four available habitat types in each arena: dry-bare (DB), dry-cover (DC), moist-bare (MB) and moist-cover (MC). The locations of the metamorphs were determined over three time periods: At Release, after 12 h, and after 36 h. A value of 7 means that seven of 10 spadefoots in a trial (n = 10 spadefoots per trial; n = 13 trials) selected the given habitat type. Closed circle = outliers, shaded box = interquartile range, dark horizontal bar = median, top horizontal bar = greatest value excluding outliers, and the bottom horizontal bar = least value excluding outliers.

Pearson's Chi-squared Tests in the lattice package in R (Sarkar 2008). We analyzed the zero-inflated, grouped, non-normal count data with mixed effects using a zero-inflated Poisson (ZIP) Generalized Linear Mixed Model (GLMM; Bolker et al. 2013) to effectively predict metamorph microhabitat selection. The ZIP model accounted for the high number (38.5%) of true zeros in the count data (Lambert 1992; Zuur et al. 2009). We used the glmmADMB package (Fournier et al. 2012; Skaug et al. 2015) with a loglink function to determine whether there was a relationship between the number of observations (i.e., number of metamorphs) within each quadrant for each trial and time period, and the three explanatory variables direction, habitat-type, and soil temperature. We attributed the random effects to our grouping (i.e., nesting) variables, time at 0 h, 12 h, 36 h and trial number 1,

TABLE 1. The mean soil temperature (°C; ± standard deviation) for each habitat type: dry-bare (DB), dry-cover (DC), moist-bare (MB) and moist-cover (MC), at three time periods at the study site in the Thompson River Valley, British Columbia, Canada: At release, at 12 h, and at 36 h. We used eight Maxim Integrated iButton Thermochron ® temperature data loggers (Model DS1921G, San Jose, California, USA) to record soil temperature every hour within the artificial enclosures.

Time Period	DB	DC	MB	MC
At release	26.1 ± 4.9	26.6 ± 4.9	25.5 ± 4.8	25.6 ± 4.6
12 h	24.5 ± 3.1	22.9 ± 1.5	21.1 ± 1.6	20.6 ± 1.8
36 h	24.6 ± 3.3	23.0 ± 1.9	21.8 ± 2.2	21.7 ± 3.1

2, 3, 4. We completed the analysis by checking the data set for overdispersion (Crawley 2002; Zuur et al 2012).

RESULTS

In total, we collected data from 13 arena trials (in four batches) and 130 animals. The mean soil temperature collected from the data loggers ranged from 20.6° ± 1.8° C and 26.6° ± 4.9° C at three time periods (i.e., At release, 12 h, and 36 h) and four habitat types (i.e., moist-cover, dry-bare, dry-cover, and moist-bare; Table 1). A strong trend towards the increase in the selection of the moist-cover habitat quadrant was seen over the duration of the study (Fig. 4).

Data on microhabitat choice from the 13 trials could be pooled for both cover type (Fisher's Exact Test, $P = 0.070$) and direction (Fisher's Exact Test, $P = 0.070$). The GVIF values for direction, cover, and soil temperature were 1.02, 1.11, and 1.09 respectively. We frequently found metamorphs (n = 130) concentrated in the different quadrants, which in turn resulted in the majority of quadrant counts containing zero animals.

The resulting analysis of the pooled data indicated that metamorph habitat choice was dependent on cover type ($X^2 = 17.82$, df = 9, $P = 0.037$) but independent of the direction of the arena ($X^2 = 7.65$, df = 9, $P = 0.619$). At release, the percentages of metamorphs that dispersed into the moist-cover, dry-bare, dry-cover, and moist-bare habitats were 26%, 25%, 24%, and 25%, respectively. After 12 h, 75% of the metamorphs were located in moist-cover habitat. After 36 h, 79% of metamorphs were located in moist-cover habitat rather than dry-bare, dry-cover, or moist-bare habitats. Metamorphs strongly preferred moist-cover habitat type ($z = 6.24$, $P < 0.001$, n = 130). All other habitat types (i.e., dry-bare, dry-cover, moist-bare), direction, and soil temperature (recorded by data loggers) were not significant (all other Ps > 0.190).

DISCUSSION

Previous laboratory experiments have shown that recently metamorphosed anurans prefer some type of vegetative cover over bare ground (not including moisture as a factor; Heinen 1993; Baughman and Todd 2007), but

this study demonstrates that newly metamorphosed Great Basin Spadefoots more specifically prefer moist microhabitats with cover. Although this response is intuitive for a metamorphic amphibian entering an arid terrestrial environment, this study is the first clear demonstration for such a preference within the metamorphic stage of the Great Basin Spadefoot. Whereas this work was performed in an arena, it seems likely that metamorphs emerging from water bodies would exhibit similar behavior in a natural setting. Observations in the field support this idea: we detected metamorphs taking refuge in visibly moist soil under plywood adjacent to an evaporating pond and within moist fissures at the edges of other ponds (Hales 2018). Our results parallel those of Weintraub (1980), who found recently metamorphosed New Mexico Spadefoot (*Scaphiopus multiplicatus*) individuals under boards, in shallow retreats, cow dung, or fissures caused by drying of pond edges during the daytime. Baughman and Todd (2007) found newly metamorphosed Eastern Spadefoots preferred habitat with forest litter (i.e., cover) in laboratory experiments in South Carolina. Walston and Mullin (2008) found that juvenile amphibians in Illinois, specifically the Small-mouthed Salamander (*Ambystoma texanum*), American Toad, and Wood Frog (*Rana sylvatica*), exhibited non-random orientation moving in the direction of forested habitat versus disturbed open areas with little canopy cover. Combined, these findings suggest that many, if not most, species of amphibians transitioning from an aquatic juvenile stage to one that uses drier upland habitat would preferentially seek out moist habitat with cover.

Garner (2012) studied upland habitat selection of Great Basin Spadefoots near their northern range limit (about 110 km northwest of the present study). Using telemetered adult animals, Garner (2012) found selection for daytime retreat sites that contained a relatively high proportion of bare ground. Superficially, this seems to suggest a very different pattern of habitat selection between adults and emerging newly metamorphosed Great Basin Spadefoots (i.e., habitat preferences might change or shift as the animals ages). The Garner (2012) study site, however, was situated within a cooler, moister ecosystem than the present study, suggesting the animals may have been more limited by heat and thus selected for bare (warmer) ground in which to bury. In 2014, average daily temperatures near the Garner (2012) study site in June, July, and August ranged from 3.9–17.9° C, 6.6–24.0° C, and 6.2–23.3° C, respectively (http://climate.weather.gc.ca/index_e.html#access). Total precipitation in June, July, and August was 36.5 mm, 49.7mm, and 17.4 mm, respectively (http://climate.weather.gc.ca/index_e.html#access). Further, the smaller surface area-to-volume ratio of adult animals may enable them to remain buried underground for long periods of time, retaining and/or absorbing soil moisture through their permeable skin (Ruibal et al. 1969). Metamorphs, being much smaller in size, likely have a lower tolerance for

dry conditions, and therefore have different habitat requirements. This study did not examine all factors (e.g., food availability, body condition) that could influence the terrestrial habitat choice of newly metamorphosed spadefoots, but it did determine that, given the conditions provided in the arena, moist habitat with cover is preferred directly following metamorphosis.

This work has important implications for the management of the Great Basin Spadefoot, particularly in the semi-arid grasslands of British Columbia. Given the results of this study and the Environment and Climate Change Canada (ECCC; 2017) recovery strategy, we recommend retaining or establishing breeding water bodies with natural or artificial cover material (e.g., plants, coarse woody debris, cover boards) situated along the immediate perimeter of the pond edge to provide shelter and moisture. Such conditions should increase the quality of suitable terrestrial microhabitats for vulnerable, newly metamorphosed spadefoots. The presence of cover features may be extended over larger areas surrounding water bodies to provide additional resources for both newly metamorphic and adult Great Basin Spadefoots as they disperse from the breeding pond (ECCC 2017; Hales 2018). Breeding ponds should be protected (e.g., fencing, enclosures) from sources of disturbance (e.g., cattle, development). Metamorph mortality caused by cattle disturbance (e.g., metamorphs trapped in hoof prints along pond edge) was observed by the author at Great Basin Spadefoot breeding ponds throughout the study. Livestock use of water bodies and riparian areas can also leave shorelines and neighboring habitat devoid of vegetation and other cover (Cragg 2007).

Technology (particularly telemetry) currently restricts our knowledge of upland habitat requirements for metamorphs of the Great Basin Spadefoot. Knowledge of the dispersal phase for these animals is particularly limited: the distance travelled from the breeding pond upon metamorphosis, daytime retreat site locations and types, and hibernation locations are virtually unknown for juvenile Great Basin Spadefoots. Inventive methods will need to be employed to collect these sorts of data. For example, Popescu and Hunter (2011) tracked movement and recorded habitat preferences of newly metamorphosed Wood Frogs using runway enclosures with pitfall traps and tracking stations. Similar methods may be needed to further our understanding of habitat requirements for spadefoots across all life-history stages.

Animal Care Committee).

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Survey for *Coccidioides* Antibodies in Blood Sera from California Sea Lions and Northern Fur Seals that Stranded Along the Coast of California (2013–2015)

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Abstract.—Coccidioidomycosis, also known as Valley Fever, infects marine mammals along the coast of California. These animals reside far from the Central Valley of California and the Mojave Desert where the fungal pathogen *Coccidioides immitis* is endemic. An agar gel immunodiffusion (AGID) assay established to screen for coccidioidomycosis exposure in humans and canines was used on blood sera collected from 136 stranded California Sea Lions (*Zalophus californianus*) of all age groups and 17 Northern Fur Seals (*Callorhinus ursinus*) between 2013 and 2015, to quantify the prevalence of exposure to the pathogen. We detected *Coccidioides*-specific antibodies in approximately 11% of California Sea Lions and 24% of Northern Fur Seal pups that were diagnosed with respiratory problems at time of admission to two marine mammal care centers. No significant difference was detected in risk of exposure to the pathogen when comparing sex, or stranding location of California Sea Lions, however, we found that yearlings of California Sea Lions did not show any evidence of seroconversion. Our results confirm that exposure to *Coccidioides* occurs among pinnipeds, likely due to strong Santa Ana winds, which carry dust from mainland California to the Pacific Ocean. Thus, disease resulting from infection with *Coccidioides* should be considered as a possible cause for stranding in pinnipeds. The risk of exposure to the pathogen can be considered equal for California Sea Lions, Northern Fur Seals, and other marine mammals that spend significant time in an area affected by mainland dust. The location of greatest exposure is difficult to determine, however, because of sea lion migration behavior and variable wind and weather patterns. The findings of this study have implications for marine mammal rehabilitation and conservation, and also for public health.

Key Words.—agar gel immunodiffusion assays; *Callorhinus ursinus*; coccidioidomycosis; immunoglobulin M; immunoglobulin G; marine mammal rehabilitation; pinnipeds; stranding; *Zalophus californianus*

Introduction

Major causes of pinniped (seals and sea lions) strandings along the coast of California include natural or manmade disasters, storm surges, El Niño events (that often result in malnutrition), human or predator related injuries, infectious diseases, and non-infectious diseases (cancer, congenital, etc.) including domoic acid poisoning (Gerber et al. 1993; Colegrove et al. 2005; Greig et al. 2005). Malnutrition lowers the functional performance of the immune system and renders animals more vulnerable to infectious diseases (Simeone et al. 2015). Stranded pinnipeds that suffer from malnutrition are often also diagnosed with pneumonia, which can be caused by various infectious agents (viruses, bacteria, fungi, and/or parasites). The cause of pneumonia, and other infectious diseases, must be identified for successful treatment, rehabilitation, and release of a rescued marine mammal.

In coastal California, pinnipeds and other marine mammals (i.e., Sea Otters [*Enhydra lutris*], dolphins, and whales) have occasionally been diagnosed with coccidioidomycosis, also known as Valley Fever, a disease caused by a soil-borne fungal pathogen (Reed et al. 1976; Cornell et al. 1979; Fauquier et al, 1996; Kirkland and Fierer 1996; Reidarson et al. 1998; Carlson-Bremmer et al. 2012). Recently, coccidioidomycosis was identified as the most common mycosis in stranded marine mammals along the central California coast (Huckabone et al. 2015; Simeone et al. 2015). Unfortunately, the disease is often diagnosed postmortem, when treatment with antibacterial drugs was unsuccessful or when supportive care of animals failed. At that stage, the disease likely disseminated throughout the body, and affected lymph nodes, lungs, and bones (Gerber et al. 1993; Shubitz and Dial, 2005; Simpson and Cornell 2018). For example, over a six-year period, 22.2% of rehabilitated pinnipeds in California presented with pneumonia of unknown origin (suspected bacterial or verminous infections). Some of these animals might have contracted coccidioidomycosis (Gerber et al. 1993) but were not diagnosed. Between 2005 and 2014, 12 California Sea Lions (*Zalophus*

californianus) rescued at The Marine Mammal Center (TMMC) succumbed to coccidioidomycosis (Appendix Table). Early detection and proper treatment of coccidioidomycosis could have improved the outcomes for these stranded animals. Understanding the potential sources of this pathogen, typically associated with dry, desert habitats, may be helpful to determine when and where testing for the pathogen may be a cost-effective diagnostic tool for marine mammal rescue efforts.

The opportunistic fungal pathogens, *Coccidioides immitis* and *C. posadasii*, both cause coccidioidomycosis. In California, *C. immitis* is highly endemic in the central and southern San Joaquin Valley and the western Mojave Desert where it can be found in non-disturbed fine particulate soils (alluvium) dominated by salt bushes (particularly Allscale, *Atriplex polycarpa,* Spiny Saltbush, *A. spinifera*, Shadscale, *A. confertifolia*, and Fourwing Saltbush *A. canescens*) and Creosote (*Larrea tridentata*), whereas *C. posadasii* is established in other areas of the Southwestern U.S., Mexico, and parts of South America (Baptista-Rosas et al. 2007; Fisher et al. 2007). Arthroconidia (dormant forms of the pathogens) can become airborne when soil is disturbed and when inhaled, arthroconidia can form spherules in the lung, potentially resulting in pneumonia and occasionally disseminating to other internal organs (Pappagianis 1980).

The migratory behavior of marine mammals hinders precise determination of where these animals may be exposed to the pathogen. California Sea Lions tend to migrate long distances along the U.S. west coast and can range from southern Mexico and Baja California, where *C. posadasii* resides, to British Columbia (considered non-endemic for *Coccidioides*). Rookeries (breeding grounds) are located almost entirely on the Channel Islands of California, western Baja California, and the Gulf of California (Aurioles et al. 1983; Laake et al. 2018). Therefore, it can be inferred that pinnipeds spend significant time in waters and rookeries exposed to mainland dust potentially containing *C. immitis* (California) or *C. posadasii* (Mexico) and are likely at risk of exposure while in these areas. Northern Fur Seals (*Callorhinus ursinus*), which have established rookeries on San Miguel Island close to the *C. immitis* endemic mainland, are at risk of pathogen exposure as well, but instead of migrating south, these pelagic animals prefer the north Pacific Ocean with another large rookery on the Farallon islands near San Francisco, California; however, most breeding grounds of this species are in the Aleutians Islands in the Bering Sea.

There are two likely sources of the pathogen: fugitive dust from offshore islands and from areas known to be endemic for *C. immitis* and/or *C. posadasii* on the mainland. Although many of the Channel Island plant communities resemble comparable communities on the California mainland, they are dominated by endemic species unique to the islands. In response to soil and climate conditions that are different from the mainland, communities on the islands form distinct plant alliances and associations such as Island Chaparral, Coastal Sage Scrub, and Grassland. *Atriplex polycarpa* and other *Atriplex* species, as well as Creosote that are adapted to saline-alkaline desert soils, can indicate a suitable habitat for *Coccidioides*, but are not established on the Channel Islands (Junak et al. 2007; Channel Island National Park. 2019. Channel Islands Plant Check List. National Park Service, U.S. Department of the Interior. Available from https://www.nps.gov/chis/planyourvisit/upload/A-Checklist-of-Vascular-Plants-all-v1.pdf [Accessed 6 October 2019]). Therefore, it seems unlikely that *Coccidioides* are an important part of the fungal community in Channel Islands soils.

The recent surge (from about 2010 to now) in land development in the California deserts for utility-scale solar energy development (USSED), in addition to fallow farmland resulting from a long-term drought, has contributed to an increase in fugitive dust and coccidioidomycosis in highly endemic areas of the pathogen in the Mojave Desert (Hector et al. 2011; Thompson III et al. 2015; Guevara et al. 2015; Grayzel et al. 2017). Recent studies in the western Mojave Desert, located north of the San Gabriel Mountains that separates the Los Angeles Basin from the Mojave Desert, revealed the presence of *C. immitis* in 29% of all soil samples collected from disturbed and non-disturbed soils at sites destined for large-scale renewable energy projects west of Lancaster, California (Etyemezian et al. 2018). Since 2011, a significant increase in coccidioidomycosis among humans has been observed in Kern (southern San Joaquin Valley) and northern Los Angeles (Mojave Desert) counties (Cooksey et al. 2017; Colson et al. 2017). Recent research also revealed an earlier onset of the spring fine dust season in the endemic area of the pathogen due to climate change (Hand et al. 2016), which likely leads to an increased risk of exposure to *Coccidioides* for animals and humans.

Strong seasonal Santa Ana winds arriving at the end of the long summer drought season (May-September/October) in California can transport pathogenic arthroconidia from dry and eroded land in the Mojave Desert to the Los Angeles Basin and beyond, putting people as well as domestic and wild animals at risk of developing coccidioidomycosis (Westerling et al. 2004; Duniway et al. 2019; Appendix Figure). Between 2013 and 2017, strong Santa Ana winds carried dust from the Mojave Desert to non-endemic areas of the pathogen along the coast and fanned several wildfires (Kolden and Abatzoglou 2018). These events increased particulate matter concentrations of 10 μm or less in diameter (PM10) in the air along the coast of California and are a growing public health issue (Black et al. 2017).

In anticipation of regular Valley Fever surveillance in the future, we aimed to obtain initial information about how common exposure to *Coccidioides* is among stranded

California Sea Lions admitted to two marine mammal care centers in California over a 3-y period. A small number of Northern Fur Seal pups was also included in this study because of unusually high numbers of strandings in 2014 and 2015. Here we present preliminary data from a series of planned tests that will continue monitoring coccidioidomycosis in California Sea Lions and Northern Fur Seals through 2021 (NOAA permit # 19706). Valley Fever monitoring would also be of interest to scientists at the National Oceanic and Atmospheric Administration (NOAA) Fisheries agency (https://www.fisheries.noaa. gov/national/marine-life-distress/2013-2017-california-sea-lion-unusual-mortality-event-california) who are investigating California Sea Lion Unusual Mortality Events (CSL UME).

METHODS

Study sites.—Pinnipeds admitted to the Marine Mammal Care Center Los Angeles (MMCCLA) in San Pedro, California, and to The Marine Mammal Care Center (TMMC) in Sausalito, California, were included in this study. The MMCCLA admits animals stranded in Los Angeles County (Malibu to Seal Beach); whereas, TMMC responds to animals stranded between San Luis Obispo County and Mendocino County. Both Marine Mammal Care centers (collectively, MMCs) identify the causes of stranding, document and diagnose stranded animals, and treat them with the intent of rehabilitation and release.

Blood serum samples.—We tested stored, frozen blood serum samples from California Sea Lions and Northern Fur Seals rescued between 2013 and 2015 for antibodies against *Coccidioides* spp. Additional information was gathered as available, including the cause of stranding, any diagnoses and analyses performed by the veterinarians at the MMCs at any stage of treatment at the site of recovery, and whether the animal was released. Information gathered from animals found dead or that were euthanized while in rehabilitation (e.g., lesions in the lung, in bones, or in inner organs) was included as well. We focused on stranded California Sea Lions and Northern Fur Seals diagnosed with a respiratory disease (suspected pneumonia) because respiratory problems can indicate coccidioidomycosis in an early stage (Pappagianis 1980; Valdivia et al. 2006). Overall, we analyzed serum samples from 136 California Sea Lions (all age groups; MMCCLA San Pedro: n = 70, TMMC Sausalito: n = 66). In addition, we included 17 stranded Northern Fur Seals (16 pups and one subadult) that stranded in 2014 and 2015 in this study.

Immunodiffusion assays.—We performed an agar gel immunodiffusion (AGID) assay for *Coccidioides* specific immunoglobulin M (IgM) and immunoglobulin G (IgG) on serum samples. Samples were frozen, aliquots were

prepared by personnel of the MMCs, and were shipped on dry ice to the Public Health Laboratory at Kern Medical in Bakersfield, California. These assays were performed following the method described in Pappagianis and Zimmer (1990). This method is identical to the method performed at the School of Veterinary Science of the University of California, Davis (UC Davis), which is used to diagnose coccidioidomycosis in humans, dogs, and occasionally marine mammals (Pappagianis 2001; Gautam et al. 2013; Schmitt and Procter 2014).

Statistical analyses.—We limited all statistical tests to California Sea Lions because the number of positive AGID assays from Northern Fur Seal pups was too small to conduct meaningful statistical analyses (n = 17, with four positive tests). Furthermore, we pooled all positive immunodiffusion assays from California Sea Lions because a positive test for either IgG or IgM indicates exposure to the pathogen. We conducted statistical analyses based on the binary variable of exposure (positive vs. negative) for either test for each animal.

We used the Log-Likelihood Ratio test for Contingency Tables (Zar 1999; McDonald 2014) to determine if AGID test results of blood serum from California Sea Lions were significantly different between individual years from 2013–2015, sexes, and age classes. We conducted post-hoc power analyses with Gpower, assuming either a moderate effect size of 0.3 or a small effect size of 0.1. Finally, we used the G-test (Woolf 1957) with Yates correction for continuity (Haviland 1990) to analyze if stranding location (by county) might be indicative of a higher risk of *Coccidioides* exposure in California Sea Lions.

RESULTS

We screened 136 serum samples from California Sea Lions admitted to MMCCLA or TMMC between 2013 and 2015 for the presence of *Coccidioides* specific antibodies IgM and IgG using AGID assays to investigate the prevalence of exposure to the pathogen. Serum from adult female sea lions comprised most samples obtained from animals rescued by MMCCLA at San Pedro (about 77%, n = 54); whereas, samples from TMMC in Sausalito had a fairly even distribution of sex and age groups. For both MMCCs combined, about 36% of the sea lions were adults and about 16% were pups (Table 1).

We tested California Sea Lions of all age groups that stranded between Los Angeles County and Mendocino County. Overall, 51% of serum samples were from animals recovered by MMCCLA in San Pedro and 49% from animals rescued by TMMC Sausalito (Table 2). Positive immunodiffusion assays for IgM ranged from about 8–14% between 2013–2015. The prevalence of positive IgG test results was low and ranged between about 2–3% in each year (Table 3). The proportions of California Sea Lions testing positive for IgM or IgG

TABLE 1. Number of stranded California Sea Lions (*Zalophus californianus*), by sex and age class, admitted to Marine Mammal Care Center Los Angeles in San Pedro (MMCCLA), California, and to The Marine Mammal Center in Sausalito (TMMC), California, between 2013 and 2015 that presented with a respiratory infection (n = 136) and which contributed to this study. Distribution % is the distribution of age classes from all animals at both Marine Mammal Care Centers.

	MMCCLA (n = 70)	TMMC (n = 66)	Distribution % (n = 136)
Females	54	34	64.71
Males	16	32	35.29
Pup	2	20	16.18
Yearling	2	18	14.71
Juvenile	0	6	4.41
Subadult	23	16	28.68
Adult	43	6	36.03

across years was not significantly different (G = 1.427, *P* = 0.499). There was also no significant difference in risk of exposure to the pathogen between sexes (with year and age groups pooled; G = 0.007, *P* = 0.935); however, we observed a significant difference between age groups and exposure to the pathogen (with years and sex pooled; G = 10.12, *P* = 0.018; Table 4).

Stranding in an area closer to the endemic regions of the pathogen (Los Angeles Basin) was also not an indicator of increased risk of exposure to the pathogen in these cases (Yates corrected G = 0.020, df = 1, *P* = 0.888). In Los Angeles County about 13% of the 70 stranded sea lions tested positive for either IgG (1.43%) or IgM (11.43%), compared to 10.6% (3.03% positive for IgG, 7.57% positive for IgM) for all other counties combined (n = 66; Fig. 1). Overall, 10.86% of animals that stranded in either area had detectable amounts of IgM or IgG.

TABLE 2. Percentage of California Sea Lions (*Zalophus californianus*) of all age classes (n = 136) and Northern Fur Seals (*Callorhinus ursinus*), with few individuals, mostly pups (n = 17), that were part of this study, sorted by stranding location (counties, from south to north). The sea lion from Stanislaus County (not a coastal county) lived in a zoo.

Counties	California sea lions % (n = 136)	Northern fur seals % (n = 17)
Los Angeles	51.47	11.77
Santa Barbara	8.82	0
San Luis Obispo	8.08	29.40
Monterey	11.03	5.88
Santa Cruz	5.88	11.77
Stanislaus	0.74	0
San Mateo	7.35	5.88
Alamedo	0.74	0
San Francisco	2.21	5.88
Marin	1.47	5.88
Sonoma	2.21	11.77
Mendocino	0	11.77
Total	100%	100%

Results of the AGID assays from blood sera of Northern Fur Seals identified four of 17 animals positive for *Coccidioides* exposure. Two animals were positive for *Coccidioides* specific IgM only, while two additional fur seals were positive for both antibodies (all pups). No fur seals were positive for IgG alone. Most of these animals stranded in San Luis Obispo County, but strandings occurred as far north as Sonoma and Marin counties. The two Northern Fur Seal pups rescued in Los Angeles County were negative for both antibodies. Overall, 24% of Northern Fur Seal pups were positive: 12% of the pups tested positive for IgM alone and 12% tested positive for both IgM and IgG. Of the four animals that showed an immune response to *Coccidioides* antigens, two stranded in 2014 and two in 2015.
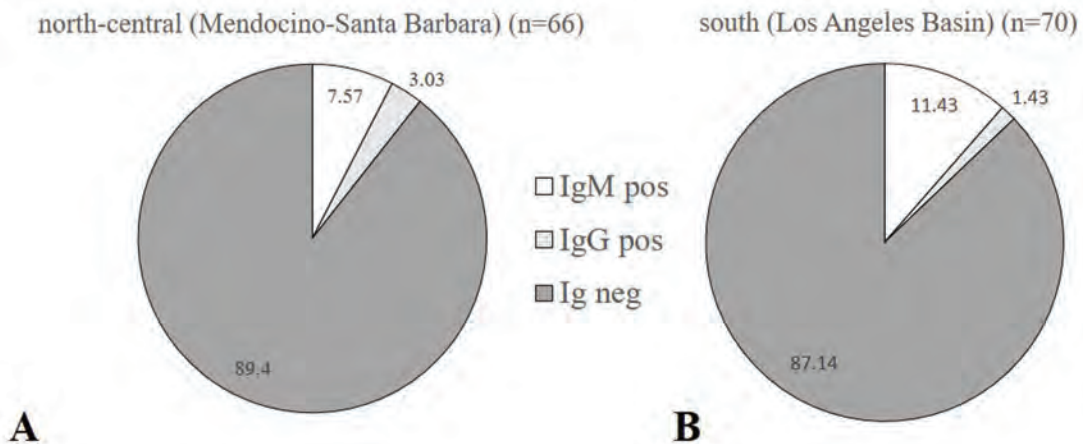

FIGURE 1. Results of agarose gel immunodiffusion assays for *Coccidioides* specific antibodies (IgM and IgG) comparing the percentage of California Sea Lions (*Zalophus californianus*) that (A) stranded in north-central counties (Mendocino-Santa Barbara County) of California and (B) the percentage of animals that stranded in Los Angeles County.

TABLE 3. Results of agarose gel immunodiffusion assays for blood serum from 136 California Sea Lions (*Zalophus californianus*) that stranded between 2013 and 2015 and which were diagnosed with respiratory problems including suspected pneumonia. Results are separated by year of stranding and by age classes. The abbreviation n = sample size.

Age classes	n	Sex		Positive AGID assays	
		Females	Males	IgM	IgG
2013					
Adult	22	21	1	3	0
Subadult/Juvenile	8	6	2	1	1
Yearling	1	0	1	0	0
Pup	5	4	1	1	0
Total n	36	31	5	5	1
Percentage	100	86.11	13.89	13.89	2.78
2014					
Adult	21	17	4	1	0
Subadult/Juvenile	16	7	9	2	0
Yearling	5	4	1	0	0
Pup	9	4	5	2	1
Total n	51	32	19	5	1
Percentage	100	61.54	37.25	9.8	1.92
2015					
Adult	7	7	0	0	1
Subadult/Juvenile	20	8	12	3	0
Yearling	13	13	0	0	0
Pup	8	3	5	0	0
Total n	49	24	25	3	1
Percentage	100	48.98	51.02	8.33	2.78
Total n (2013–2015)	136	87	49	13	4

DISCUSSION

Our results support earlier studies showing that California Sea Lions and Northern Fur Seals living along the coast of California or those migrating through this area are at risk of exposure to *C. immitis*, a fungal pathogen that may be transported to coastal areas via strong Santa Ana winds from endemic areas in the Mojave Desert (Simeone et al. 2015; Huckabone et al. 2015). This fugitive dust can be transported to the South Coast Air Basin that includes areas in Los Angeles, Orange, Riverside, and San Bernardino counties. Extreme weather events have been associated with cases of coccidioidomycosis in traditionally non-endemic areas of the pathogen, including one in a gorilla in the Sacramento Zoo (Pappagianis and Einstein 1978). The Northridge earthquake in 1994 is another event that transported dust and *Coccidioides* arthroconidia north to Sacramento, California, and into Oregon (Schneider et al. 1997). We cannot exclude the possibility that *C. immitis* persists or grows in soils near pinniped rookeries, however, possibly inoculated by transported dust. Muhs (1983) and Muhs et al. (2007, 2008) provided evidence from soil and dust minerology studies supported by Landsat imagery that aeolian dust originating from the Mojave Desert deposits regularly on the Channel Islands

where pinniped rookeries are established (Laake et al. 2018). Muhs (1983) estimates that dust deposit rates on San Clemente Island range between 28–31 $g/m^{2/y}$, reflecting ongoing and accelerated erosion of soil surfaces disturbed by human activity in the Mojave Desert, which has increased in recent years (Sprigg et al. 2014; Tong et al. 2017). Urban et al. (2018) suggest that the minimum dust emission (PM10) from the Mojave Desert currently ranges between 3–8 Tg/y. While *Atriplex polycarpa* and other plants associated with *Coccidioides* in the San Joaquin Valley and the Mojave Desert are not established on the Channel Islands (Junak et al. 2007), other *Atriplex* species do occur. Recent efforts, however, to detect *Coccidioides* in coastal soils north of Cambria (San Luis Obispo County near a Northern Elephant Seal rookery), near Ventura (Ventura County) and on Santa Catalina Island (several different inland and coastal ecosystems) have been unsuccessful (Antje Lauer, unpubl. data). Therefore, fugitive dust from mainland sources either in California or Baja California in Mexico (Laniado-Laborin 2007) appears to be a more probable exposure risk for marine mammals along the coast of California.

Although dust generated in the Mojave Desert and transported to coastal areas has not been investigated for the presence of *Coccidioides* arthroconidia, indirect evidence suggests that fugitive dust generated in endemic areas of

TABLE 4. *Coccidioides* prevalence in California Sea Lions (*Zalophus californianus*) that were part of this study across years (log-likelihood test; sex and age class pooled) 2013–2015, between sexes (year and age-class pooled, with Yates correction), and between age classes (years and sex pooled). Independent rather than joint comparisons were made due to sample size considerations; given that none of the comparisons were significant, no corrections for non-independence have been included. Statistical power was calculated post-hoc under assumptions of a moderate effect size of 0.3 and a small effect size of 0.1 to provide a reasonable range. The abbreviations n = sample size, df = degrees of freedom, G = G*Power, *P* = *P*-value.

Comparison	Categories	Percentage Positive	over-all n	df	G	*P*	power (0.3)	power (0.1)
between years	2013	16.7	36	2	1.43	0.49	0.89	0.17
	2014	11.7	51					
	2015	8.2	49					
between sexes	female	10.3	87	1	0.007	0.94	0.94	0.21
	male	12.3	49					
between age groups	pup	18.2	22	3	10.02	0.02	0.85	0.14
	yearling	0	20					
	subadult/juvenile	15.9	44					
	adult	10.0	50					

the pathogen has caused disease elsewhere (Pappagianis and Einstein 1978; Flynn et al. 1979; Schneider et al. 1997; Fisher et al. 2012). Dust potentially carries *Coccidioides* arthroconidia to offshore islands and could therefore be a source of infection for pinnipeds and other marine mammals. Recently, Polymerase Chain Reaction (PCR) fragments of *Coccidioides* ribosomal DNA from dead California Sea Otters (*Enhydra lutris nereis*) found in coastal counties were shown to be 99% related to a 460 bp fragment obtained from soils west of Lancaster, California (98% query coverage, GenBank nucleotide database at the National Center for Biotechnology and Informatics [NCBI]: GenBank Accession # KY306699; Huckabone et al. 2015; Colson et al. 2017), which is consistent with the hypothesis that fugitive dust from the Antelope Valley of California is a source of exposure and possible infection with *Coccidioides* in traditionally non-endemic areas.

We detected no significant differences in exposure risk for California Sea Lions between 2013 and 2015. Because many adult females remain near the Channel Islands year-round, we anticipated an increased risk of pathogen exposure for these females relative to subadult and adult males, which roam to areas that are not as impacted by fugitive dust from the mainland (far out into the Pacific Ocean or further north), but our data show that both sexes had similar exposure risks (rates of seroconversion). When comparing age groups, however, we found that yearlings which made up about 16% of the sea lions did not show any evidence of seroconversion to the pathogen based on AGID assays. This led to the observed significant difference between age classes. A larger sample size may provide some insights as to whether older animals have higher risk of exposure to *Coccidioides* and if pups benefit from maternal antibodies for at least some time. We also expected to see a higher risk of exposure for animals stranding in counties closer to endemic areas of the pathogen, such as San Luis Obispo and Los Angeles counties, but instead we found no significant difference among locations. This could indicate similar risks of exposure across coastal counties or could instead reflect the migratory behavior of the animals meaning that stranding location may not be a useful proxy for location of exposure.

The dramatic increase in reported coccidioidomycosis incidence among humans residing in coastal California counties since 2016 is a concern (California Department of Public Health [CDPH] 2019. Infectious diseases by disease, county, year, and sex. CDPH. Available from https://data.chhs.ca.gov/dataset/infectious-disease. [Accessed 19 September 2019]) and should also alert animal conservationists to consider potential increases among wildlife populations, including pinnipeds (Appendix Figure 2). A comparison between locations in which sea lions presented with *Coccidioides* exposure and location in which humans presented with coccidioidomycosis could be attempted in the future with more expanded sampling and analysis of sea lion serum. A significant increase in *Coccidioides* titers in screened sea lions or other pinnipeds, such as Harbor Seals (*Phoca vitulina*) might serve as a warning for human exposure to the pathogen. Indeed, in 2017 and 2018, TMMC reported additional cases of coccidioidomycosis among California Sea Lions and MMC San Pedro identified several suspected cases that responded to antifungal treatment (pers. observ.). In addition, it is interesting to note that California Sea Lions are being used as sentinels for ocean and public health serving as indicators for domoic acid poisoning and other health concerns (Bossart 2011).

AGID is the most commonly employed serological test used to diagnose exposure to *Coccidioides* in humans and animals but has not been used on pinniped blood samples on a regular basis. The sensitivity of this test when performed on canine blood serum samples was determined to be 87% for immunoglobulin G and 46% for immunoglobulin M (Gunstra et al. 2019). The sensitivity and specificity of AGID assays are generally similar to antibody Enzyme Immuno Assays (EIAs) using infected canines (Holbrook et al. 2019). In another study investigating canines, AGID for IgG and IgM was

found to be 100% sensitive (100%) confirming results by histopathology (Johnson et al. 2003). Determining antibody titers can help distinguish acute from chronic disease; however, quantitative titration of antibodies does not always correlate with clinical disease in dogs or other animals (Greene and Troy 1995; Shubitz and Dial 2005; Shubitz 2007; Graupmann-Kuzma et al. 2008; Burgdorf-Moisuk et al. 2012). Therefore, AGID results should be evaluated with caution and results from ancillary diagnostic studies (e.g., from cytology, histopathology, culture, or PCR) should be included when available to confirm the diagnosis. Although this serology method has not yet been validated in sea lions or other marine mammals to correctly diagnose coccidioidomycosis, the test might serve as an indicator of exposure to the pathogen. According to Pappagianis and Zimmer (1990), the concentration of antibodies in sera from human patients generally decreases to undetectable levels for most patients with resolved infections. Therefore, patients with detectable anti-coccidioidal antibodies likely have recent exposure, illness or reactivation, or have chronic active disease.

Based from results from this and past studies, it is reasonable to assume that an AGID assay detects *Coccidioides* specific IgM in pinnipeds and indicates coccidioidomycosis in an early stage of the disease (acute infection). As disease progresses, IgM is replaced with IgG, which can indicate late or chronic disease process, or low levels may indicate acquired or waning immunity from previous exposure to the pathogen (Pappagianis and Zimmer, 1990). In our study, the titer of anti-*Coccidioides* antibodies in pinniped blood sera was not determined but would be necessary to distinguish actual disease (and the severity of it) from cell mediated immunity that indicates exposure to the pathogen (Pappagianis and Zimmer 1990). Paired serology studies would improve our ability to interpret titers (acute versus convalescent), but this method was beyond the budget of this study. Recently, Chow et al. (2017) developed an enzyme immunoassay (EIA) to detect *Coccidioides* specific antibodies in mammalian species in general by validating this method on dogs and mice. Neither method has been tried on blood sera from pinnipeds. A careful validation of the AGID and EIA as an indicator of both disease and exposure to *Coccidioides* would increase the confidence of using this method as a diagnostic tool for pinnipeds.

Because cell mediated immunity is the key mechanism of defense against coccidioidomycosis (Graupmann-Kuzma et al. 2008), early diagnosis of coccidioidomycosis, followed by treatment with an antifungal agent and supportive care that strengthens the immune system of an animal, should improve recovery from the disease. Delayed diagnosis of the disease poses the risk of dissemination, environmental contamination, unsuccessful rehabilitation, and potential death. For example, coccidioidomycosis was not considered as an underlying cause of stranding for three sea lions (one subadult female [case # 11977, stranded in Santa Cruz County] and two juvenile males (case #12549, stranded in San Mateo County, and case #11502, stranded in San Luis Obispo County) that were euthanized or died in treatment in the same year (2011 and 2012); all animals were IgM positive for *Coccidioides*. Approximately 1/3 of California Sea Lions with positive immunodiffusion assays (either IgM or IgG) in this study died during treatment or were euthanized.

Use of a quantitative titer in addition to AGID assays for IgM and IgG to distinguish acquired immunity to the pathogen from acute or chronic infection, particularly if the animal does not respond to treatment for more common causes of pneumonia (bacteria and parasites) and the exclusion of other causes of respiratory disease, may enhance detection of active infections with *Coccidioides*. This strategy will increase the chance for successful rehabilitation and avoid expensive and ineffective treatments with potentially hazardous side effects (Butler et al. 1964; Graupmann-Kuzma et al. 2008). The focus of this study was on blood serum from animals that presented with respiratory problems, but it is important to acknowledge that not all pinnipeds that present with respiratory issues are infected with *Coccidioides*. Future studies should continue to examine blood serum from all stranded animals and determine if respiratory problems are a good indicator to consider coccidioidomycosis as an underlying cause of stranding in pinnipeds as it is in human patients.

We plan to continue our work with pinniped blood sera to 2021 (under NOAA permit 19706), to relate seroprevalence or rate of detected cases of coccidioidomycosis to environmental changes, such as El Niño events (in 2015 and 2016), effects of future droughts, and increases in fugitive dust from the endemic mainland due to increases in soil disturbance and dust events. Analysis of this long-term dataset may provide correlations between human and pinniped exposure and possibly disease incidence, which could provide conservationists with an early warning of expected increases in strandings due to coccidioidomycosis. We are confident that these early efforts will help to establish a continuous monitoring project for disease incidence among stranded California Sea Lions, which are the most numerous rescued pinnipeds in the MMCs, as indicators for *Coccidioides* exposure risk for other marine mammals along the coast of California. Our data are an important baseline to help achieve this goal. Our work will contribute to improved outcomes in pinniped rehabilitation and conservation efforts by raising awareness of a disease that is often misdiagnosed and overlooked in regions that are not endemic for *Coccidioides* but may be impacted by fugitive dust emission from endemic areas of the pathogen. Furthermore, assessment of the health status of stranded marine mammals not only provides valuable information concerning care of these animals but also

provides information about existing and emerging health problems in wild populations and environmental health.

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ANTJE LAUER is a Professor at California State University Bakersfield (CSUB) having taught there since 2007. She received her M.S. in Biology from the University in Oldenburg, Germany, her Ph.D. in Microbiology from the University of Kiel, Germany, and obtained additional research experience at the University of North Carolina at Chapel Hill (Department of Marine Sciences) and at James Madison University in Virginia (Department of Biology). At CSUB, she teaches courses in Environmental and Medical Microbiology and is currently conducting research on Valley Fever in California. (Photographed by Nancy Lauer).

LAUREN PALMER has been the Staff Veterinarian at Marine Mammal Care Center Los Angeles (California) since 2005. She has contributed to several publications on Elephant Seal (*Mirounga angustirostris*) anatomy, pinniped parasitology, and infectious diseases of stranded marine mammals. She received her Veterinary degree from the University of Minnesota, Minneapolis, Minnesota, her Master of Public Health degree from the University of Iowa, Iowa City, Iowa, and an undergraduate degree in Environmental Science from California State University, Northridge. (Photographed by Lauren Palmer).

Carl Kloock has been at California State University Bakersfield since 2001 and is a Professor of Biology. He received his B.S. (1989) in Zoology and M.S. (1991) in Ecology from the University of California, Davis, and his Ph.D. in Evolution, Ecology and Behavioral Biology from Michigan State University, East Lansing (2000). He is a Behavioral Ecologist and works primarily on problems of aggressive mimicry, particularly in arachnids. (Photographed by Carl Kloock).

Heather Liwanag is an Assistant Professor of Biology at Cal Poly San Luis Obispo, California. She earned her B.S. in Biology from the University of California, San Diego, a Ph.D. in Ecology & Evolutionary Biology from the University of California, Santa Cruz, and was a post-doctoral fellow at the University of California, San Diego, and the California State University, Northridge. She is interested in the physiological adaptations of animals to their environment, and the evolutionary processes involved in those adaptations. (Photographed by Center for Teaching, Learning & Technology at Cal Poly San Luis Obispo).

Tenaya Norris has been involved in marine mammal research and conservation since 2004. She completed her Master's degree at Moss Landing Marine Laboratories, Sausalito, California, in 2013. For her thesis research, she studied the efficacy of translocations as a conservation strategy for endangered Hawaiian Monk Seals (*Monachus schauinslandi*). In her current role as Marine Scientist at The Marine Mammal Center, Sausalito, California, Tenaya collaborates with scientists in the USA and Mexico to study the abundance, distribution, foraging behavior, habitat use, and health of pinnipeds in the California Current System, particularly Guadalupe Fur Seals (*Arctocephalus townsendi*). (Photographed by Brandon Stone).

Birgitte McDonald is an Assistant Professor at Moss Landing Marine Laboratories, Moss Landing, California, where she has been teaching courses on Ecology and Physiology of Marine Vertebrates and Scientific skills since 2015. She obtained her Ph.D. at the University of California, Santa Cruz, and conducted postdoctoral research at Scripps Institution of Oceanography, San Diego, California, and Aarhus University in Denmark. As a Physiological and Behavioral Ecologist, Gitte investigates adaptations that allow animals to survive in extreme environments. (Photographed by Inge McDonald).

Jazmine Mejia Muñoz received her B.S. in Biology from California State University, Bakersfield. She is currently a graduate student in the Environmental Science Master of Science program at California State University, Monterey Bay. She is a Louis Stokes Alliance for Minority Participation (LSAMP) Scholar, a Maximizing Access to Research Careers Undergraduate Student Training in Academic Research (MARC U-STAR) Scholar, a Sally Casanova Scholar, and a Fuhs Fellow. (Photographed by Anna Jacobsen).

Catherine Mulcahy received a B.S. degree from Worcester State University, Worcester, Massachusetts, in 2014 and was a Research Assistant at The Marine Mammal Center, Sausalito, California, from 2015–2016. (Photographed by Catherine Mulcahy).
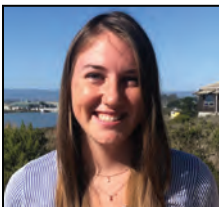
Shawn Hannah is a graduate student at Moss Landing Marine Laboratories (MLML), Moss Landing, California, working toward her M.S. in Marine Science, and she received her B.S. in Biology at Cal Poly San Luis Obispo, California. At MLML, Shawn is examining the plasticity of the dive response of Northern Elephant Seals (*Mirounga angustirostris*) and how it is affected by fine-scale dive behavior. (Photographed by Lauren Cooley).

Shawn Johnson joined The Marine Mammal Center in Sausalito, California, in 2012. His role as Director of Veterinary Science is to guide the science team's study of marine mammal health issues around cancer in sea lions, toxic algae poisoning effects, and Harbor Seal (*Phoca vitulina*) health analysis, among others. He has also worked with the Alaska Sealife Center, Alaska's Fish and Wildlife Services, the National Marine Mammal Laboratory, the Oiled Wildlife Care Network and the Wildlife Health Center at the University of California, Davis. (Photographed by Elena Graham © The Marine Mammal Center).

Appendix Table. Information about California Sea Lions (*Zalophus californianus*) diagnosed with *coccidioidomycosis* postmortem between 2005 and 2014 at The Marine Mammal Center (TMMC) in Sausalito, California. Age classes are based on straight length of body (Laake et al. 2016). Abbreviations for age classes are Y = yearling, J = juvenile, S = subadult, and A = adult, and for sexes are M = male and F = female.

Case #	Date	County	Age class	Sex	Result	Cause of death
6519	30 April 2005	San Luis Obispo	Y	M	died in treatment	disseminated coccidioidomycosis, malnutrition
6897	5 June 2006	Santa Barbara	J	M	died in treatment	disseminated coccidioidomycosis, malnutrition
7036	17 September 2006	San Luis Obispo	A	F	died in treatment	disseminated coccidioidomycosis, malnutrition
7830	13 August 2008	Santa Cruz	A	F	euthanized	severe coccidioidomycosis, domoic acid toxicity, malnutrition
8108	4 May 2009	Santa Cruz	A	F	died in treatment	disseminated coccidioidomycosis, domoic acid toxicity
9739	2 July 2010	San Luis Obispo	Y	F	died in treatment	peritonitis (unknown), malnutrition
9748	4 July 2010	Monterey	S	M	died in treatment	peritonitis (unknown)
9991	7 July 2011	Santa Cruz	S	F	died in treatment	disseminated coccidioidomycosis
10054	13 August 2011	San Luis Obispo	A	F	died in treatment	disseminated coccidioidomycosis, sepsis
10279	3 June 2012	Monterey	S	M	died in treatment	disseminated coccidioidomycosis, sepsis (perforation)
10630	21 April 2013	Monterey	J	M	euthanized	disseminated coccidioidomycosis, myocardinal infarction
10700	14 August 2013	Monterey	J	M	euthanized	disseminated coccidioidomycosis
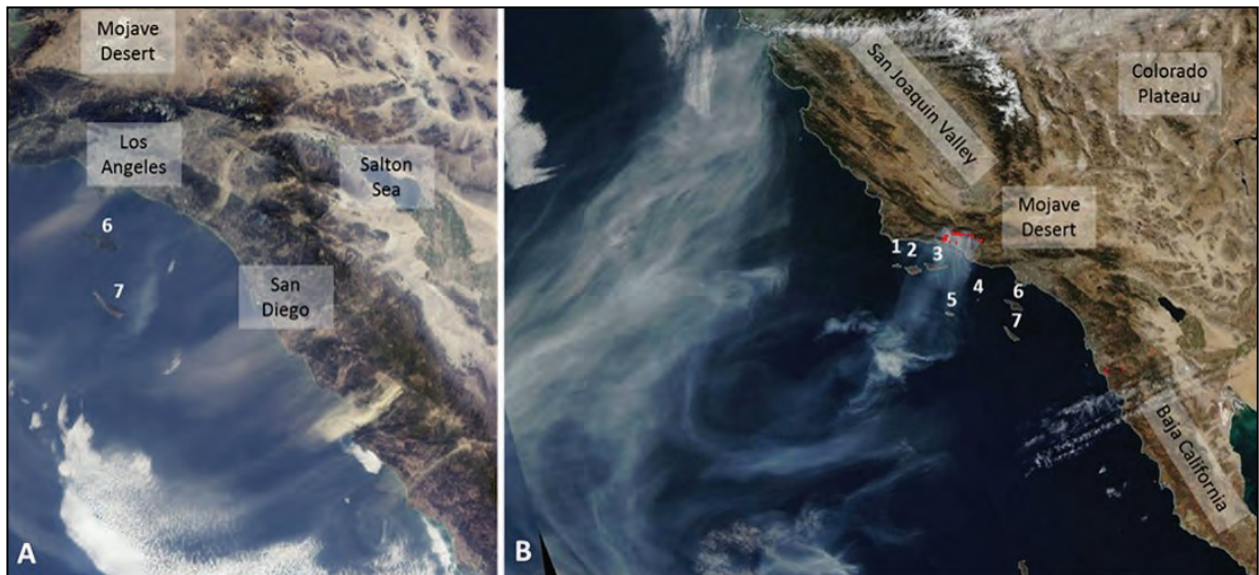
Appendix Figure 1. (A) Example of dust transport from mainland California during Santa Ana conditions on 9 February 2002 (modified from NASA image, Multi-angle Imaging SpectroRadiometer [MISR] 2019). (B) Sparked by strong Santa Ana winds, inland dust and smoke from the Thomas Fire (13 December 2017) was carried several hundred miles into the Pacific Ocean (modified from NASA image, NASA Worldview Application operated by the NASA/Goddard Space Flight Center Earth Science Data and Information System [ESDIS] project). Actively burning areas were detected by Moderate Resolution Imaging Spectroradiometer [MODIS]; thermal bands are displayed in red. The numbers refer to Channel Islands of California. 1: San Miguel Island, which supports major California Sea Lion (*Zalophus californianus*) and Northern Fur Seal (*Callorhinus ursinus*) rookeries, 2: Santa Rosa Island, 3: Santa Cruz Island, 4: Santa Barbara Island, 5: San Nicolas Island (major California Sea Lion rookeries), 6: Santa Catalina Island, 7: San Clemente Island.
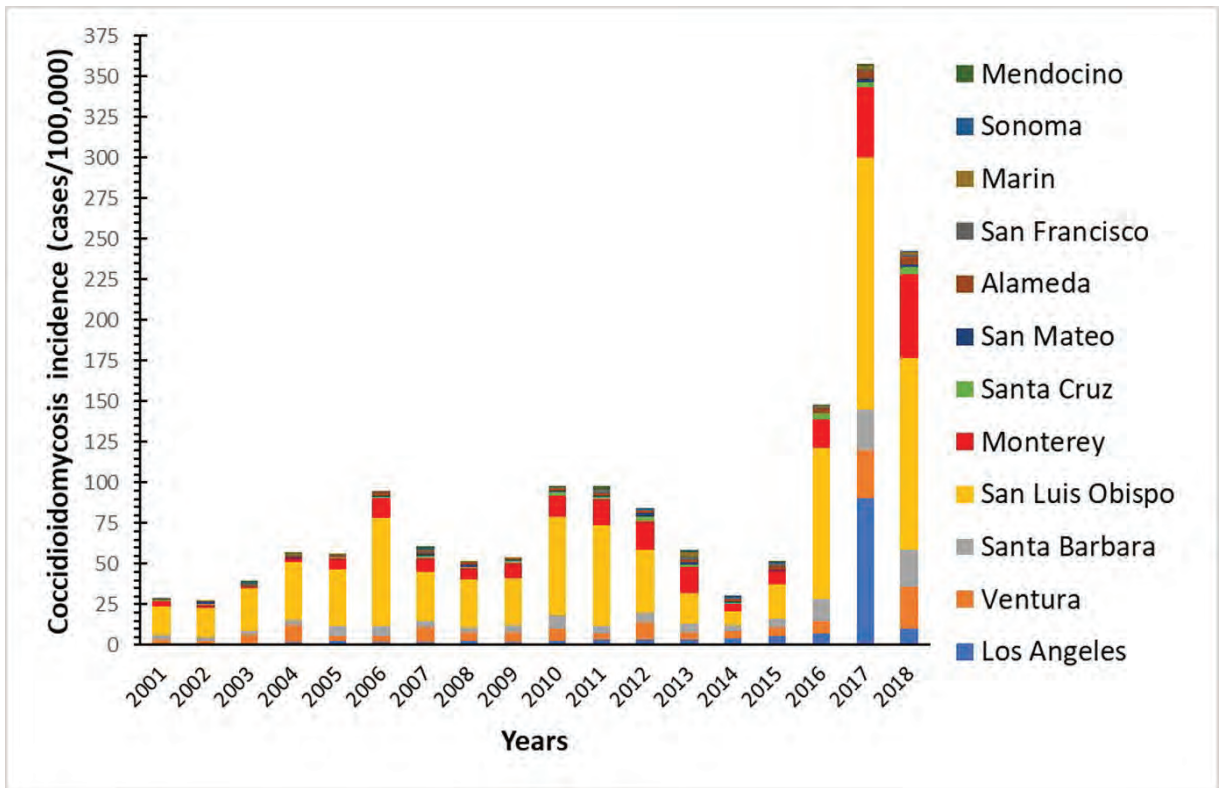
Appendix Figure 2. Reported incidence of coccidioidomycosis in humans residing in coastal California counties between 2001 and 2018. Counties are shown from south (bottom) to north (top; data obtained from: https://data.chhs.ca.gov/dataset/infectious-disease).

Program Chair: **Matthew Bettelheim**. AECOM.

Attendance: 616 participants including 153 students and 57 Early Career Professionals.

Plenary Theme: Death And Taxas: Extinction and Speciation During the Anthropocene.

Plenary Speakers: Dr. Alexis Mychajliw, Paleobiologist, La Brea Tar Pits of Los Angeles, California; **Dr. Peter H. Bloom**, Zoologist, V.P. , Bloom Biological, Inc.; **Dr. Ben Sacks**, Professor, University of California, Davis; **Tom Maloney**, Director of Conservation, Revive & Restore.

Keynote Address: Adapt or Die: Changes in who we serve and who we are. **Dr. Jennifer Malpass**, Bird Banding Lab Biologist at the USGS Patuxent Wildlife Research Center.

Awards Bestowed

- The Raymond F. Dasmann Award for the Professional of the Year went to **David J. Germano**.
- The Conservationist of the Year Award went to the **Yosemite Conservancy**.
- The Chapter of the Year Award went to the **Nevada Chapter**.
- The James D. Yoakum for Outstanding Service and Commitment to The Western Section of The Wildlife Society went to **Cynthia Perrine** and **Richard Burg**.

Student Presentation Awards

Oral Presentations:

1st Place: **Leila S. Harris**, Assessment of the status of the Townsend's big-eared bat in California, University of California, Davis

2nd Place: **Connor M. Wood**, Broad-scale monitoring for broad-scale challenges: bioacoustics and the California spotted owl, University of Wisconsin-Madison

3rd Place: **Trinity N. Smith**, Elucidating patterns of bat species occupancy across a disturbed landscape in California's Central Valley, Humboldt State University

1st Place: **Diana Munoz**, Feral horses disrupt greater sage-grouse lekking activity in the Great Basin, US Geological Survey/UC Davis

2nd Place: **Matthew Delgado**, Ornament morphology and soft tissue and skeletal correlates in Aleutian cackling geese (*Branta hutchinsii leucopareia*), Humboldt State University

3rd Place: **Shelby P. Moshier**, Climate change vulnerability is phylogenetically clustered for most of California's birds, California State University, Fresno

2019 Western Section Membership

Regular: 670	Life-Full: 35
Student: 181	Life-Partial: 5
Early Career Professional: 118	Honorary: 1
Retired: 54	Supporting: 14
Contributing: 1	

Total: 1079

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