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Black Skimmer (*Rynchops niger*) flying with elegant wingbeats at the Hayward Regional Shoreline (Tern Town), San Francisco Bay, California.
(Photographed by David Riensche).

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NOTES

REDISCOVERY OF THE COLUMBIA PLATEAU POCKET MOUSE (*PEROGNATHUS PARVUS*) AT THE NORTHERN EXTENT OF THE RANGE OF THE SPECIES

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Abstract.—**Peripheral populations are frequently exposed to marginal conditions; however, adaptations to these suboptimal conditions makes these populations important for genetic diversity and conservation. We detected the Columbia Basin Pocket Mouse (*Perognathus parvus*) at the extreme northern extent of their range, in a valley where they were last reported in 1949. If the species persists in this valley, aridification of the area due to climate change may allow *P. parvus* to eventually replace the Western Deer Mouse (*Peromyscus sonoriensis*) as the dominant cricetid rodent in the semi-arid grasslands of British Columbia.**

Key Words.—British Columbia; grassland; Great Basin Pocket Mouse; Heteromyidae; *Perognathus parvus*; population; range.

The Columbia Plateau Pocket Mouse (*Perognathus parvus*; previously the Great Basin Pocket Mouse) ranges from southeastern California northwards through Nevada, Utah, Idaho, Oregon, and Washington, USA, before reaching its northern limit in southern British Columbia, Canada (Nagorsen 2005). At the northern limits, these mice are restricted to threatened arid grassland habitats (https://a100.gov.bc.ca/pub/eswp/esr.do?id=16091). They require dry, sandy soils for burrowing and sandbathing (Kritzman 1974) and tend to be replaced by Western Deer Mice (*Peromyscus sonoriensis,* previously *P. maniculatus*) at higher altitudes (O'Farrell 1975). In British Columbia, the species is considered Special Concern (Ramsay and Nagorsen 2024) although it has not yet been assessed at the federal level. Historic records show *P. parvus* in three principal areas in the province (Fig. 1): (1) in the south Okanagan Valley, which continues east to the Kettle Valley and west to the Similkameen Valley; (2) in the north Okanagan Valley; and (3) in the Thompson River Valley.

The arid grasslands in southern British Columbia are contiguous with grasslands in Washington, and some species associated with this ecosystem have been shown to have gene flow across the U.S.-Canada border (Schmidt 2019). The north Okanagan and Thompson regions, however, are more isolated. The arid grassland habitat in the north Okanagan is bordered to the south by the urban sprawl of Kelowna, which is suspected to segregate this population from those of the south Okanagan (Nagorsen 2005). To the west of the Okanagan lies the Thompson Valley that also contains arid grasslands. The two valleys are geographically isolated from one another, and the Thompson Valley ecosystem likely has existed as a habitat island since the Holocene Climate Optimum (about 5,500 years before present), when mid to high elevation grasslands transitioned to forest, blocking habitat corridors in the region (Mathewes and King 1989). Many of the species in the Thompson Valley reach their northern limit here and so are more limited by marginal climatic conditions than populations in more southern habitats. This combination of genetic isolation and marginal conditions leaves fringe populations with increased risk of extirpation from these areas (McDonald and Brown 1992; Williams et al. 2010). The conservation value of these populations, however, often is high due to their strong contribution to the overall genetic diversity of a species, as fringe populations tend to have experienced differing selection pressures than central populations within the range of a species (Lesica and Allendorf 1995; Williams et al. 2010).

Until our study, the last records of pocket mice in the Thompson Valley and the north Okanagan were 1949 and 1951, respectively (Nagorsen 2005). A small mammal live-trapping program in the grasslands surrounding the city of Kamloops (1997-present) has failed to detect pocket mice, along with several other short-term studies overlaid on the long-term sites (unpubl. data). Additional trapping in the valley also has been undertaken in communities outside of Kamloops with similar results (Hales 2011).

In the summer of 2019, a small mammal live-trapping inventory (Pereira 2019) captured 20 individual Columbia Plateau Pocket Mice in Kalamalka Lake Provincial Park, in the North Okanagan just outside the city of Vernon (Latitude 50.207895, Longitude -119.256038) within grassland and dry, open forest habitat. These captures were 4.2 km from the nearest known capture location (1937) and 7.7 km from a specimen caught in 1951.

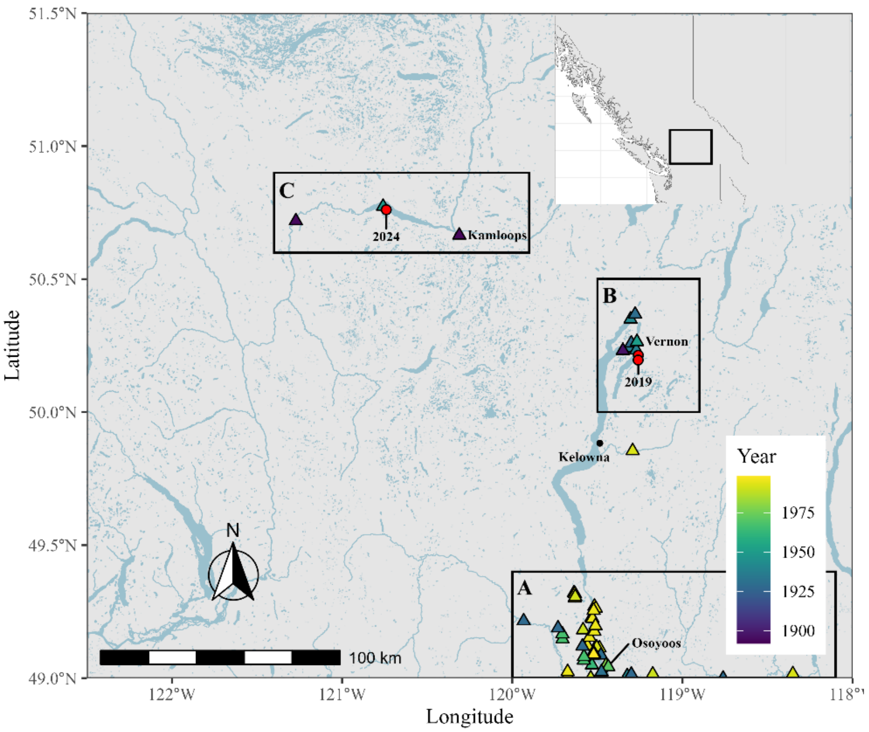

FIGURE 1. Known locations of the Columbia Plateau Pocket mouse (*Perognathus parvus*) in British Columbia, Canada. Populations are defined by three primary areas: (A) the south Okanagan Valley, near the town of Osoyoos, (B) the north Okanagan Valley, near the town of Vernon, and (C) the Thompson Valley, near the city of Kamloops. Until recently, *P. parvus* had remained undetected in the Thompson and north Okanagan Valleys since 1949. Recent trapping (red points) in the north Okanagan (2019) and the Thompson Valley (2024) re-establishes the presence of the animal in these historic locations.

One of these two historic capture locations has since been developed and now consists of residential and commercial developments (pers. obs.).

On 28 August 2024, we caught a single Columbia Plateau Pocket Mouse during a short-term live trapping study within an isolated habitat patch near Kamloops Lake, close to the community of Savona (Latitude 50.751280, Longitude -120.840732). The individual was a non-reproductive adult female (Fig. 2) with a mass of 19.0 g, a zygomatic width of 12.45 mm, a pes length of 22.5 mm, and an ear length of 7 mm. The trapping session consisted of three nights of pre-baiting with sunflower seeds, whole oats, and apple pieces followed by three consecutive nights of trapping. The trapping grid consisted of 40 Longworth-style traps spaced 15 m apart in a 4 × 10 grid. Earlier in the same summer (18–20 June), an identical trapping session that we conducted at the same location failed to produce any captures of pocket mice. Other animals we caught in the same trapping session as the pocket mouse at this location were 18 Western Deer Mice and four Yellow Pine Chipmunks (*Neotamias amoenus*).

Our 2024 capture location was within 2 km of the 1949 pocket mouse observation (specimen CM 46420 in the Carnegie Museum of Natural History Collection, Pittsburgh, Pennsylvania). The trap location was on a south-facing slope, situated in a shrub-grassland ecosystem with an historic average annual precipitation of 320 mm (Wang et al. 2016). Common plant species at the site included Bluebunch Wheatgrass (*Pseudoroegneria spicata*), Needle-and-Thread Grass (*Hesperostipa comata*), Big Sagebrush (*Artemisia tridentata*), Rubber Rabbitbrush (*Ericameria nauseosa*), and Prickly Pear Cactus (*Opuntia fragilis*). Bluebunch Wheatgrass has been noted as a dominant species in Columbia Plateau Pocket Mouse habitat in the Okanagan (Sullivan and Sullivan 2008).

The southern interior of British Columbia is expected to become hotter and drier in coming years (Smith 2011; Prugh et al. 2018; Xu et al. 2021). Research has shown that the ubiquitous Western Deer Mouse tends to be outcompeted in hot, dry valley bottoms by pocket mice (O'Farrell 1975; Maida 2020; Melaschenko and Hodges 2020). As a result, pocket mice may benefit from both the climatic shifts and the reduced competition from the Western Deer Mouse. This pattern is supported by several studies suggesting that rare species may benefit from climate change, as they become more competitive against current dominant generalists (Jiang et al. 2013; Prugh et al. 2018).

The persistence of the pocket mouse in a *Peromyscus*-dominated community at the periphery of its range suggests the potential for range expansion into the arid valley bottoms in the Thompson Valley, mirroring their distribution in the South Okanagan and Washington (O'Farrell 1975; Maida et al. 2020; Melaschenko and Hodges 2020). Previous inventory work in the south Okanagan indicates that pocket mice reach much higher

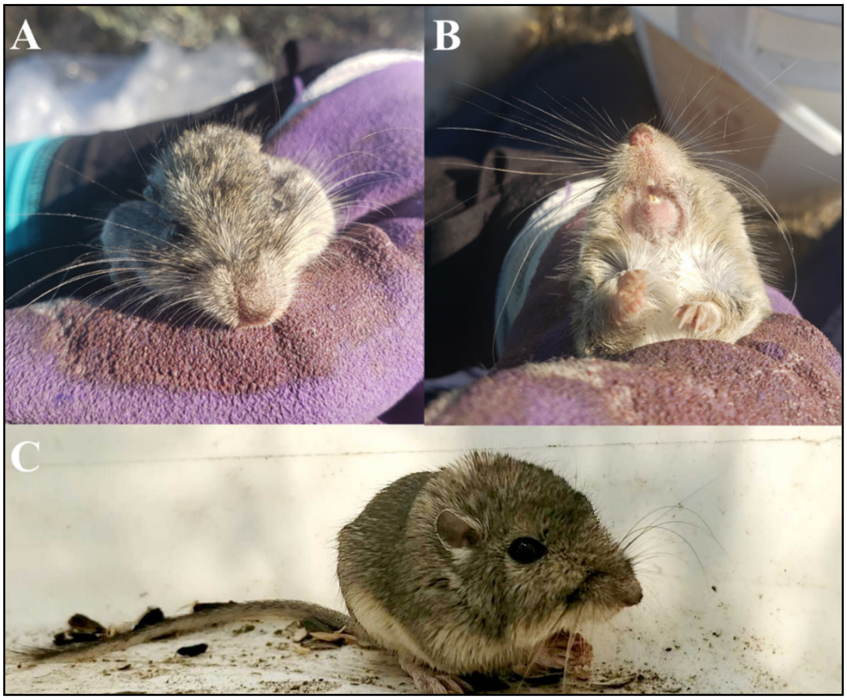

Figure 2. Photographs displaying (A and B) the cheek pouches and body and (C) tail shape of the individual Columbia Basin Pocket Mouse (*Perognathus parvus*) caught south of Kamloops Lake, British Columbia, Canada, 29 August 2024. (Photographed by Kara Atkinson).

density in natural arid grassland/sagebrush habitat than in dry forests (e.g., Ponderosa Pine, *Pinus ponderosa*) or abandoned fields and tends not to be present in agricultural land, such as orchards (Sullivan and Sullivan 2006, 2008). These authors also observed that pocket mice have poor dispersal success and highly specific habitat requirements (Sullivan and Sullivan 2008), suggesting that disturbed habitat could present significant barriers for the conservation and success of this species, as it has for other species within this genus (Brehme 2023).

The site of our recent observation re-establishes and verifies the presence of the Columbia Plateau Pocket Mouse in the Thompson Valley. This now represents the most northern known location of this species, 180 km north of the nearest published observation records. Our observation, along with the recent detections of the species in the North Okanagan, will significantly change the estimated range of these animals in Canada, a metric important to the assessment of a species at risk (Committee on the Status of Endangered Wildlife in Canada [COSEWIC] 2015; IUCN Standards and Petitions Committee. 2024. Guidelines for Using the IUCN Red List Categories and Criteria. Version 16. Available from: https://www.iucnredlist.org/ [Accessed 27 November 2024]). Additional inventory work is required in southern British Columbia to determine whether connectivity exists between the Thompson Valley population and the known locations of the animal in the south and north Okanagan. The arid grassland ecosystem of southern British Columbia is itself considered threatened (Austin et al. 2008; Williams 2015), and with increased pressure through human development, obtaining a clearer picture of the distribution of the Columbia Plateau Pocket Mouse is warranted.

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Karl W. Larsen attended the University of Victoria, British Columbia, Canada, for his undergraduate degree in Biology and Geography. This led into an M.Sc. degree with thesis work on the northern champion of the reptile world, the Common Garter Snake (*Thamnophis sirtalis*). He switched gears for his Ph.D. at the University of Alberta, Edmonton, Canada, focusing on the reproductive success of female Red Squirrels (*Tamiasciurus hudsonicus*), and an Industrial Post-Doctoral stint with Alberta-Pacific Forest Industries that took him into the realm of wildfire ecology. As a faculty member at Thompson Rivers University, he has supervised over 40 graduate students on a range of conservation-oriented projects, ranging from the movement ecology of pill bugs, the nesting ecology of Northern Goshawks (*Accipiter gentilis*), to the use of habitat patches by Asian Elephants (*Elaphus maximus*). Smaller vertebrates such as rodents and herpetofauna, however, continue to be a long-standing interest. (Photographed by Hailey Wynnyk).

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HABITAT USE AND BURROW ARCHITECTURE OF THE ENDANGERED SAN BERNARDINO KANGAROO RAT (*DIPODOMYS MERRIAMI PARVUS*): IMPLICATIONS FOR CONSERVATION

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Abstract.—**Suitable habitat is critical for the survival and reproductive success of subterranean mammals, with burrow sites playing a key role in shelter, predator evasion, food storage, and environmental regulation. For endangered San Bernardino Kangaroo Rats (*Dipodomys merriami parvus*), burrow construction in suitable locations within alluvial fan sage scrub habitat of California is essential. Here we investigate the habitat use and burrow architecture of *D. m. parvus*, whose range has been dramatically reduced by habitat loss. We collected field data through burrow casting, structural analysis, and habitat surveys. Results indicate that burrows are typically located in open, sandy areas under sparse shrub cover, with entrances that minimize soil displacement and provide escape from predators. Root cohesion likely played a key role in soil stability, as most of the burrows were adjacent to vegetation, and only small amounts of silt were measured in the soil. Burrows varied in complexity, with shallower depths that may result from relatively recent site disturbance. Anecdotal observations of burrow sharing between females and independent offspring suggest natal philopatry. Conservation efforts should prioritize open, sandy habitats with low silt in river washes, minimizing surface impacts, and maintaining adequate buffers around burrow entrances. Our study provides the first detailed examination of *D. m. parvus* burrowing ecology, offering valuable guidance for habitat management and the preservation of suitable burrowing sites for this endangered species.**

Key Words.—burrow architecture; burrowing behavior; soil suitability; habitat suitability; natal philopatry.

INTRODUCTION

Availability of suitable habitat is a key determinant of survival and reproductive success for many animal species, shaping their ability to find shelter (Swan et al. 2009), evade predators (Lima and Dill 1990), and access food resources (Halliday and Blouin-Demers 2014). For subterranean mammals, appropriate burrowing habitat is especially critical, as burrows not only provide shelter during periods of rest, but they offer protection from predators (Lacey et al. 2000), serve as storage for food reserves (Randall 1993), are used to raise offspring (Hoogland 1995), aid in conserving body moisture, and regulate microclimate, enabling thermoregulation and survival in extreme environments (Reichman and Smith 1990; Riddell et al. 2021). As a result, understanding the habitat characteristics that an animal uses for burrowing is a key component of habitat suitability and critical for management of at-risk burrowing species.

The architecture of rodent burrows is influenced by various factors. Soil characteristics, for instance, significantly impact burrow dimensions among burrowing mammals. Hard soils, such as clay, are more energy-intensive to excavate (Reichman and Smith 1990; up to 9.5 times more than sandy loam soils; Lin et al. 2017) but tend to support more complex burrow systems (Laundre and Reynolds 1993). In contrast, sandy soils without cohesive silt and clay are prone to collapse, and thus require some type of bio-reinforcement such as root cohesion (Kinlaw 1999), biocementation (Akin et al. 2024; Tirkes et al. 2024), or compaction (Akin et al. 2024) to be suitable for burrowing. Soil moisture appears to play an important role in the depth of a burrow with deeper burrows found in soils with deeper soil moisture (Bienek and Grundmann 1971). The complexity of a burrow system is also thought to be related to its function: species that primarily use burrows for shelter and raising offspring tend to construct simpler burrows, while species that also store food often build more complex ones (Reichman and Smith 1990). Additionally, the age of a burrow may affect its architecture, as long-occupied burrows can become progressively longer and deeper over time (Fitch 1948; Smith and Gardner 1985).

Kangaroo rats (*Dipodomys* spp.) typically construct complex burrows (Vorhies and Taylor 1922; Culbertson 1946; Anderson and Allred 1964) with multiple entrances, creating a network of tunnels and chambers (Kenagy 1973; Reichman and Smith 1990; Randall 1993). Because kangaroo rats are scratch-diggers, using their claws to loosen the soil (Eisenberg 1963; Nikolai and Bramble 1983; Price 1993; Siciliano Martina et al. 2023), optimal burrow sites are typically located in sandy well-drained soils that are stable yet easy to excavate, allowing for long-term burrow maintenance (Kenagy 1973; Nikolai and Bramble 1983). Burrow placement can also depend on vegetation cover, which provides food resources, shading, and protection from predators (Kenagy 1973; Gerald Braden and Robert McKernan, unpub. report).

The San Bernardino Kangaroo Rat (*Dipodomys merriami parvus*) is listed as Endangered by both California state (https://wildlife.ca.gov/Data/CNDDB) and federal (U.S. Fish and Wildlife Service [USFWS] 1998) agencies. The species was historically found in Riversidean alluvial fan sage scrub in the floodplains and adjacent upland habitat at the base of the San Gabriel, San Bernardino, and San Jacinto mountain ranges in San Bernardino and Riverside counties (USFWS 2024). Primarily due to habitat loss associated with development, mining, and water management, it was estimated that the range of the species was reduced by 96% at the time of federal listing (USFWS 1998). Currently, it is patchily distributed with only three remaining populations, each having extremely small effective population sizes (Hendricks et al. 2020). Recovery of the species depends on conserving remaining high-quality habitat and improving the suitability of low to medium quality habitat (Chock et al. 2020; USFWS 2023). *Dipodomys merriami parvus* is solitary and primarily granivorous, and like other *D. merriami* spp. (Leaver and Daly 2001; Leaver 2004), they are thought to store seeds in pit caches rather than in larders within their burrow systems. Currently, we know almost nothing about shelter use, selection of habitat for burrows, or burrow architecture in the species.

Here we describe habitat characteristics of *D. m. parvus* burrowing locations and the architecture and use of their burrows. We quantified these observations as part of a mitigation project aimed at minimizing impacts on kangaroo rats during site remediation for heavy metal contamination (Deborah Wilson and David Allison, unpubl. report). Resident kangaroo rats were removed and relocated from a mitigation area before site remediation began. A better understanding of *D. m. parvus* burrowing ecology may help inform habitat restoration and management strategies for the long-term conservation of the species.

METHODS

Study site.—We conducted burrow casting during spring 2022 on U.S. Bureau of Land Management lands in Highland, California, about 200 m north of the current path of Plunge Creek within the Santa Ana River wash (Latitude 34.104, Longitude -117.181, 393 m elevation). The site was historically part of the Santa Ana River and Plunge Creek alluvial fan complex and thus has sandy fluvial soils. From 1945 to mid-2009, the area was used as an open-air recreational shooting range. Buildings were removed from 2012–2013, leaving the alluvial fan sage scrub habitat 9–10 y to reestablish (Mikael Romich, pers. comm.) before the start of our study with early stage sage scrub, bare ground, and nonnative grass dominating. No other kangaroo rat species were documented on the site.

Burrow architecture.—We searched for kangaroo rats and documented burrow ownership throughout the footprint of the mitigation area (16.2 ha). To do this, we first live-trapped all kangaroo rats on the site using Sherman live-traps (7.62 × 7.62 × 30.48 cm; model XLKSD, H.B. Sherman Traps, Inc., Tallahassee, Florida, USA) with modified shortened doors to avoid tail injury. We spaced traps 10 m apart in grids or long lines and we opened and baited traps before dusk with sterilized millet seed and checked traps at midnight and dawn, closing them during the dawn check. We weighed, determined the sex, inspected for reproductive condition, and marked all kangaroo rats with a Passive Integrated Transponder (PIT) tag (HPT8 8 mm FDX-B, Biomark, Inc., Boise, Idaho, USA), and then we released them at the point of capture. We documented the nearest open burrow entrances of the appropriate size (e.g., with an approximately 5.0–6.3 cm entrance; Kenagy 1973). Following trapping, we used night vision goggles and remote cameras set in front of potential *D. m. parvus* burrows to confirm ownership, as determined by observing kangaroo rats entering and exiting repeatedly during the night. Once burrow ownership was determined, we trapped the kangaroo rat and removed it from the area for relocation to a new site. Following removal, we verified that the burrow was unoccupied for 24 h via camera trap images before casting the burrow.

We created casts of all unoccupied kangaroo rat burrows in situ using plaster of Paris (Reynolds and Wakkinen 1987; Laundre 1989; Laundre and Reynolds 1993; Tschinkel 2010; Dentzien-Dias and Figueiredo 2015). We poured plaster into the burrow with a funnel and a hose until the chamber was full, as indicated by mushrooming of plaster at the burrow entrance. Once the plaster hardened, we took photos to document locations of the burrow entrances. We then excavated the cast by removing layers of soil from above the casts to maintain their morphology and determine their depth and direction. If an uncast tunnel or chamber was encountered, we paused excavation, filled the opening with plaster, and waited for it to harden before continuing. Occasionally, a tunnel remained unfilled with plaster due to its uphill trajectory. In such cases, we carefully shaved off soil layers to access the tunnel from above. If we were unable to access the tunnel from above, we used polyurethane expanding foam to fill the open tunnel. We began excavation of all casts at a burrow entrance and continued until the cast ended, met another cast, or opened to the surface. We captured only one lactating female and did not cast her burrow; instead, we carefully excavated her burrow by hand to remove her unweaned offspring, and then we took measurements of tunnels and chambers.

For each burrow system (n = 10), we used a measuring tape to document the greatest burrow depth (distance from the ground surface to the burrow floor), the greatest tunnel length (the greatest distance across a series of connected tunnels), the total length of the burrow system (including all tunnels), the total number of openings, and the dimensions (width and height) of each entrance. From photos and casts, we documented the orientation of burrow entrances assigned to eight compass directions: north, northeast, east, southeast, south, southwest, west, and northwest, and used the Rayleigh test for circular uniformity to determine whether orientation differed from a uniform (random) distribution (Torres et al. 2003). We measured the distance from each burrow entrance to the basal stem of the nearest shrub, the number of shrubs directly over the excavated burrow system, and the number of shrubs within 1 m of the excavated burrow.

Habitat surveys.—We conducted ocular habitat surveys (Cheryl Brehme et al., unpubl. report) at eight of the 10 burrows in a 10 × 10 m plot centered on each burrow entrance prior to casting. At each plot we visually assessed the percentage ground cover (< 10 cm) of the following variables: (1) bare ground; (2) bare sandy soil; (3) non-native grass; (4) forbs; (5) shrubs, woody debris/leaf litter; (6) cactus; (7) native bunchgrass; and (8) inhospitable cover (e.g., boulders, concrete, gravel or paved roads). Additionally, we measured shrub cover at the crown (e.g., > 10 cm) to better characterize the extent of shrub canopy.

We measured soil compaction at the same eight burrows using a penetrometer (Model #15585, Dickey-John Corporation, Auburn, Illinois, USA) with a 76.2 cm length probe and a 1.27 cm diameter tip. We recorded the depth at which the penetrometer read 2,068 kPa (300 psi), which is the pressure roots cannot penetrate (Aase et al. 2001), and may limit kangaroo rat burrowing. We measured compaction at 25 locations evenly spaced throughout each 10 × 10 m habitat survey plot centered on a burrow entrance. We also collected soil samples from each excavated burrow following burrow casting to determine the soil particle size. Each sample consisted of approximately 5.0–6.5 kg of soil. This material surrounded the burrow and is assumed to be representative of the material excavated by *D. m. parvus* to create the tunnel. Soil particle size analysis was conducted by the Eurofins Calscience laboratory (Tustin, California, USA) using laser light scattering. Their analysis categorized particles into seven grain size classes ranging from silt and clay (< 0.0625 mm) to gravel (> 2 mm). Although sparse cobbles (> 64 mm in diameter) were observed near some burrows, and in one case (burrow 10) the tunnel passed around cobbles, they were not part of the excavated material.

RESULTS

We found that all *D. m. parvus* burrows were excavated in alluvial fan sage scrub, with no burrows documented in areas dominated by nonnative grass (Fig. 1). The burrows can be highly complex (Fig. 2), with 1–4 entrances, multiple chambers, some which were terminal, blind laterals (tunnels that do not end in a chamber), T-junctions (one tunnel intersects another at a 90° angle) and bifurcations (tunnel splits at an acute angle; Fig. 2, Table 1). Five of the 10 burrow systems had tunnels that terminated just below the surface of the

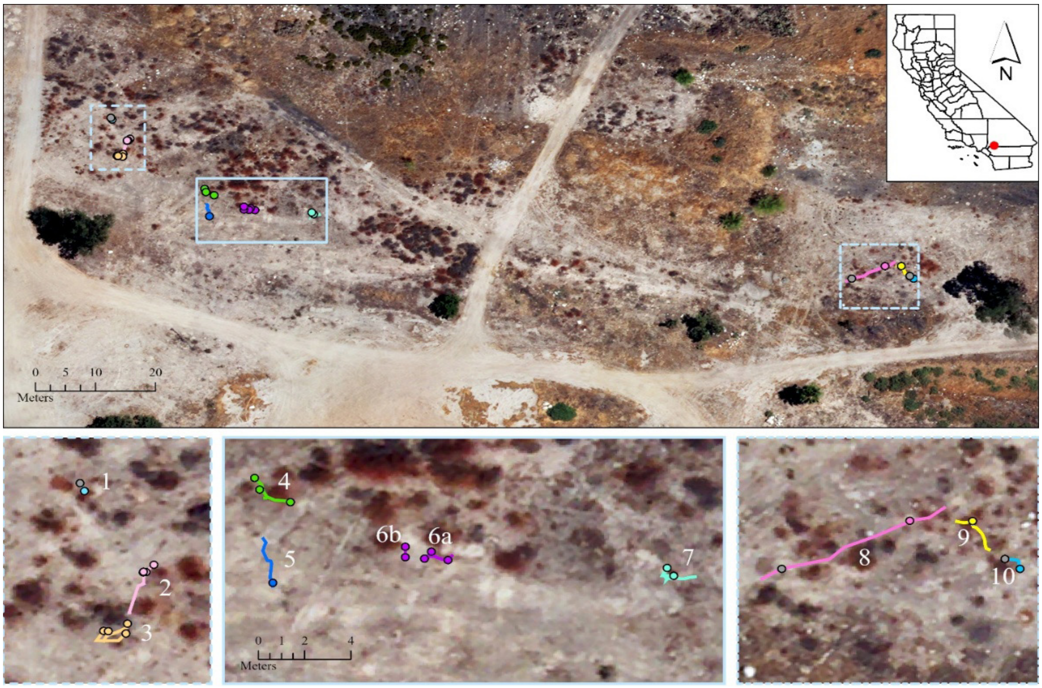

FIGURE 1. A map of the 10 burrow systems of San Bernardino Kangaroo Rats (*Dipodomys merriami parvus*) in Highland, California (red dot on map of California). The three inset panels are at the same scale as each other and show the burrows in greater detail. Lines represent tunnels and dots represent burrow entrances, with grey dots indicating plugged entrances.

Figure 2. Example photograph of burrow systems of San Bernardino Kangaroo Rats (*Dipodomys merriami parvus*) in Highland, California (left: Burrow 3; right: Burrow 2) showing plaster casts after burrow systems were excavated. Burrow entrances of each system are labeled 1–4. Inset photographs depict (a) chamber, (b) blind lateral, (c) T-junction, and (d) bifurcation. The shrubs over the burrows were trimmed during excavation; these and the surrounding shrubs are Deerweed (*Acmispon glaber*). The distance between the two burrow systems was 70 cm.

ground. Two burrow systems (1 and 6b) were relatively small and likely newly initiated or temporary refuges (i.e., subsidiary burrows, Tappe 1941) used for quick escape. Burrow entrances were roughly circular and often placed under or near shrubs with tunnels dug directly under shrubs (shrub canopy ranged from about 0.25 to 1.5 m in diameter; Table 1), and we found roots embedded in several of the plaster casts when exposed. Most burrows were oriented toward the north, but the distribution did not differ from a uniform distribution ($\bar{R}$ = 0.20, P = 0.359; Fig. 3). Similar to other *D. merriami* spp., little to no soil was piled up in the vicinity of the entrance

Table 1. Characteristics of 10 burrow systems of San Bernardino Kangaroo Rats (*Dipodomys merriami parvus*) on Bureau of Land Management property in Highland, California, excavated during spring 2022. All measurements are in centimeters. An asterisk (*) indicates that the system had a plugged burrow entrance not included here or a tunnel that ended <10 cm below the surface. A plus sign (+) indicates the orientation for one burrow entrance was not documented. Abbreviations are GD = greatest depth, LLT = length of longest tunnel, TSL = total system length, NE = number of entrances, OBE = orientation of burrow entrances, MWE = mean width of entrances, MHE = mean height of entrances, MDEBS = mean distance of entrances to base of nearest shrub, NST = number of shrubs over tunnels, NSMD = number of shrubs < 1 m to burrow system, and SD = standard deviation.

Burrow ID	GD	LLT	TSL	NE	OBE	MWE	MHE	MDEBS	NST	NSMD
1*	10.2	128.3	156.2	2	E,W	8.1	8.4	55.3	1	3
2*	35.6	327.7	396.2	4	N,N,W,SW	6.4	5.9	15.2	2	8
3	35.5	254.0	458.5	4	SE, SW,N,N	6.1	6.8	27.8	4	5
4	35.5	231.1	271.2	3	NW, SE, SW	8.7	9.3	31.3	2	3
5	17.7	306.1	306.1	1	N	4.3	4.2	320.0	0	1
6a	16.5	201.9	226.1	3	E,N,E	8.3	7.6	16.9	3	5
6b	10.2	58.4	58.4	2	N,S	6.0	6.8	83.3	0	2
7*	31.8	222.9	329.6	2	E,N	6.9	6.4	41.5	1	2
8*	27.3	894.7	1078.2	2	NE, N	8.9	10.2	33.0	0	6
9	21.6	252.1	268.0	1	N	6.7	6.0	30.5	1	6
10*	19.7	109.2	138.4	2	S	7.0	6.4	31.8	1	6
Mean	23.8	271.5	335.2	2.6		7.0	7.0	46.8	1.5	4.5
SD	9.8	222.4	271.9	1.4		1.4	1.7	61.3	1.3	2.2

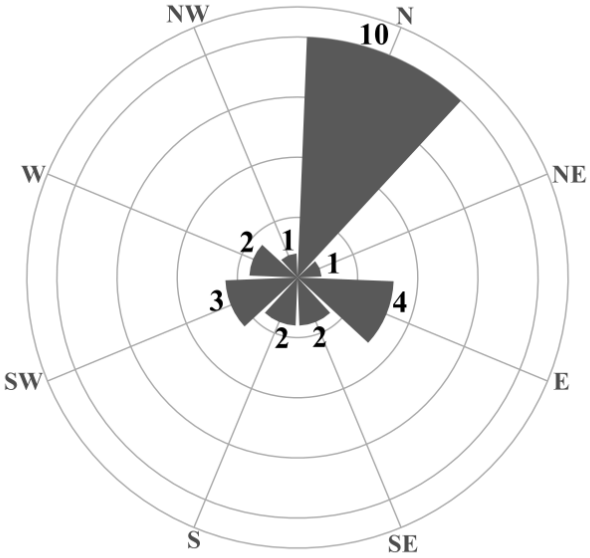

Figure 3. Orientation of 25 burrow entrances across 10 burrow systems of San Bernardino Kangaroo Rats (*Dipodomys merriami parvus*) in Highland, California. The number on each bar denotes the number of entrances in each direction. Although there were more entrances oriented towards the north, this was not a significant deviation from a random distribution.

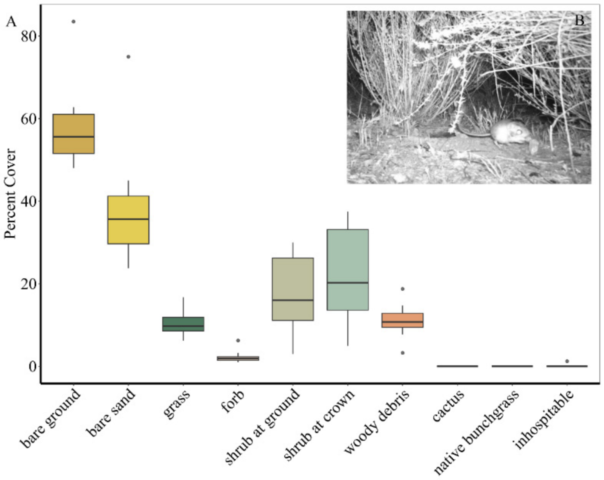

Figure 4. (A) Vegetation ground cover (< 10 cm) at eight burrow systems of San Bernardino Kangaroo Rats (*Dipodomys merriami parvus*) in Highland, California, measured on 10 × 10 m plots centered on the observed burrow entrance prior to excavation. The box plots depict medians (horizontal lines) and interquartile ranges, (IQR; boxes). Whiskers extend to the extreme values of the data or 1.5 × IQR from the center, whichever is less. Dots represent outlier values that fall outside of the whiskers. (B) Camera trap image of a *D. m. parvus* to the right of a burrow entrance on bare ground under shrub cover.

(Monson and Kessler 1940). During observations we saw *D. m. parvus* scatter hoarding, and we found no seed caches or nesting material in the excavated burrow systems.

We documented three adult females sharing burrow systems with offspring. One was a natal burrow with unweaned pups (Burrow 5). We trapped the other two adult females in the same trap locations as recently weaned independent offspring. We designated these offspring as young-of-the-year based on weight and pelage color, and we determined them to be independent based on the reproductive condition of the suspected mother (i.e., nipples had returned to normal following lactation). Based on trapping results, camera trap videos, and focal observations of burrow use, these two females each appeared to share multiple burrow systems with their weaned offspring: one female used Burrows 4 and 6a/b with a single female offspring, and one female used burrows 8, 9 and 10 with one male and one female offspring). There was a mean distance of 6.32 m between any two shared burrow systems.

Habitat surveys indicated that *D. m. parvus* burrows were located in habitat with open bare ground or open sand with shrub canopy and little grass, woody debris, or forb cover (Fig. 4). The dominant shrub present at the site was Deerweed (*Acmispon glaber*), which is a fast growing early successional species that grows in well-drained soils (https://research.fs.usda.gov/treesearch/57245). Soil at the site was relatively compacted. Although the soil compaction measurements had a wide range (Fig. 4), the median depth to 300 psi was only 5.3 cm. In addition, 96% of the soil compaction measurements were < 24 cm, the average maximum depth of the tunnels. These results suggest that the 300 psi threshold used for root growth is not a threshold for *D. m. parvus* burrowing. The median grain size for soil samples was 0.5 mm, just at the cusp between medium and coarse sand (Fig. 5). Five of the burrows (1, 2, 3, 4, and 7) were dominantly

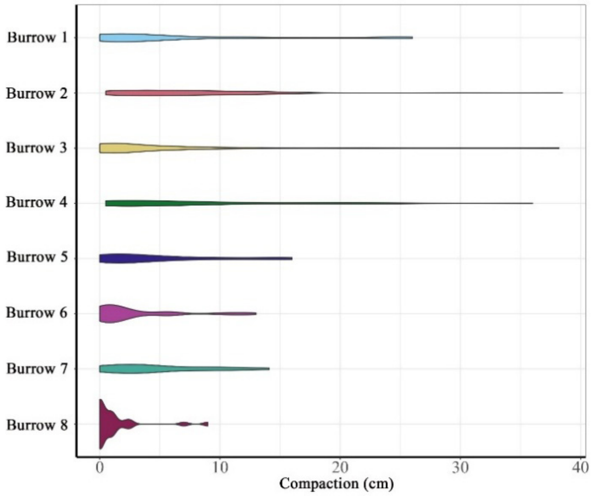

Figure 5. Soil compaction measurements in pounds per square inch (psi) across a 10 ×10 m square centered on the observed entrance prior to excavation at eight burrow systems of San Bernardino Kangaroo Rats (*Dipodomys merriami parvus*) in Highland, California. Depth to 300 psi was measured in centimeters; smaller compaction values are the most compact, and larger values are less compact (greater depth to 300 psi).

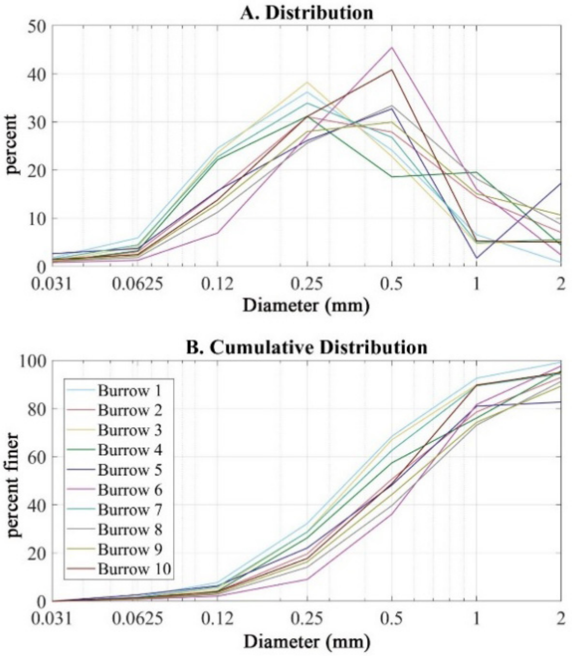

Figure 6. Grain size distribution of material removed during burrow excavation of burrow systems of San Bernardino Kangaroo Rats (*Dipodomys merriami parvus*) in Highland, California. Excavated sediment included silt and clay (< 0.0625 mm), very fine sand (0.0625–0.125 mm), fine sand (0.125–0.25 mm), medium sand (0.25–0.5 mm), coarse sand (0.5–1.0 mm), and very coarse sand (1.0–2.0 mm).

medium sand, four burrows were dominantly coarse sand (6, 8, 9, 10), and one burrow (9) had similar coarse and medium sand percentages. Silt was rare in the burrow samples, comprising an average of 1.4% of the samples. Silt made up < 2.7% of the material in all the sediment samples, suggesting that silt is not present in sufficient amounts to increase the soil strength. In addition, a cryptogamic soil crust, which is observed elsewhere in the alluvial fan, was not present (Burk et al. 2007; Brian Root, unpubl. report).

DISCUSSION

Understanding the habitat requirements of endangered species is critical for recovery, but with low numbers remaining in the wild, opportunities for filling information gaps are often rare. Our study adds to the growing body of knowledge on the habitat use of the endangered *D. m. parvus*. We found that burrow systems were located in areas with bare ground or open sand, with entrances often situated adjacent to or beneath shrub canopy. This suggests that while the soil surface remains largely unvegetated, burrow entrances may be positioned in locations where overstory vegetation provides cover. There was no significant pattern to burrow entrance orientation, which in other species have been found to be associated with wind and sun direction (Torres et al. 2003). Given the small sample size in our study, however, it remains possible that burrow orientation to the north may play a role in thermal regulation or reduced flooding as the current path of the Plunge Creek is to the south. Burrow entrances were often under shrub canopy cover, which may help mitigate these environmental factors.

There was a great deal of variation in burrow lengths, depths and complexity within the 10 burrows cast. Burrow length and depth are typically correlated with body size in rodents (Van Vuren and Ordeñana 2012). When comparing these metrics across kangaroo rat species for which burrow characteristics have been documented, no clear pattern emerged. *Dipodomys merriami parvus* appears to have longer main tunnels (mean main burrow length = 311 cm) than both Tipton's Kangaroo Rat (*D. n. nitratoides*), which has approximately the same body size, and Heermann's Kangaroo Rat (*D. heermanni*), which is substantially larger (*nitratoides*: mean burrow length = 182 cm; *heermanni*: mean burrow length = 161 cm), although some burrows of *D. n. nitratoides* reached 350 cm in length (Germano and Rhodehamel 1995).

The burrows of *D. m. parvus* at this disturbed site were fairly shallow (greatest depth = 10.2–35.6 cm, mean = 23.8 cm) compared to burrows of other *D. merriami* spp. (greatest depth 175 cm; Bienek and Grundmann 1971; Kenagy 1973). Given the high energetic cost of excavating soil (Reichman and Smith 1990), it would be advantageous for kangaroo rats to construct burrow systems that are only as long and deep as necessary to meet basic needs. Soil serves as an effective insulator, with temperatures below depths of 30–40 cm remaining largely unaffected by daily fluctuations in aboveground temperatures (Chappell and Bartholomew 1981). As a result, rodents that burrow deeper than approximately 40 cm are unlikely to experience additional thermal benefits.

Significant ground disturbance from building removal in 2012–2013 suggests that these burrows were < 10 y old. Rodent burrow depth has been shown to be correlated with burrow age (Reichman and Smith 1990). In kangaroo rats, burrow excavation may take years (Tappe 1941) with burrow systems often used by multiple generations (Best 1972). Our results provide evidence of solitary occupancy of burrow systems by *D. m. parvus* except when females are raising offspring. We documented two females sharing burrows with their presumed independent offspring after weaning. These results are consistent with natal philopatry, or the retention of offspring in natal home ranges past the age of independence from parents (Armitage 1981; Jones 1984), as documented in other kangaroo rat species (Jones 1984, 1993; Shier and Swaisgood 2012). It is possible that *D. m. parvus* burrows in areas undisturbed for longer periods may be deeper than those documented here.

Soil cohesion from biologic crust and finer soils (e.g., fine/medium sand and silt/clay) did not play a large role in stabilizing the soils at our study site. The shrub roots observed in the burrow casts at this site are a possible source of additional soil strength (Kinlaw 1999). Tirkes et al. (2024) used a soil stability model to demonstrate that kangaroo rat burrows in the Sonoran Desert are likely unstable without additional strength from cohesion provided by biologic crusts. Their study observed relatively larger burrow diameters (12 cm compared to 7 cm in this study) and finer soils (median grain size, d50 = 0.16 mm versus 0.5 mm in this study). These factors likely result in less stable soils at their Sonoran Desert site in the absence of cohesion. Further investigation throughout the remaining range of *D.m. parvus*, particularly at upland sites that likely differ in soil composition, is needed to understand the burrow architecture and soil strength and cohesive properties of soils that support *D.m. parvus* burrows. Such studies would provide insights into the conditions that promote their stability and inform conservation and restoration practices.

We provide the first information on the subterranean habitat use of the endangered *Dipodomys merriami parvus*. For a species with habitat that is heavily impacted by human activities, research from even a single site can provide important information for minimizing impacts on below-ground habitat. Burrows were longer (up to about 900 cm for a single main tunnel; > 1,000 cm total system length) and shallower (as shallow as 10.2 cm at greatest depth) than expected compared to similarly sized species of kangaroo rat. If project fencing is needed to reduce impacts to the species, we recommend that fencing is constructed a minimum of 10 m from identified burrow entrances to ensure all entrances remain on the same side of the fence. The shallow burrow depths also suggest surficial impacts of off-road vehicles could collapse burrow systems in sandy wash habitat. Burrows were in habitat comprised primarily of open bare ground, though shrub canopy may be important for buffering entrances from sun and wind or providing shelter from predators. Additionally, the relationship between roots, soil strength and cohesion, and burrow architecture needs to be investigated across multiple sites with varying vegetation and soil characteristics. Our research emphasizes the need to preserve open, sandy areas that include shrub cover to support burrowing in this species, and additional studies are needed to identify critical burrowing habitat across the remaining range of *D.m. parvus*.

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NOTES

WIDESPREAD USE OF HIGHWAY GUARDRAILS AND OTHER ANTHROPOGENIC FEATURES BY THE COLORADO CHECKERED WHIPTAIL (*ASPIDOSCELIS NEOTESSELATUS*)

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Abstract.**—The Colorado Checkered Whiptail (*Aspidoscelis neotesselatus*) is a triploid, parthenogenetic lizard with a native range endemic to southeastern Colorado. It occurs in a variety of habitats, often associated with slopes. In its natural habitat, these lizards spend much of their time actively foraging often followed by a period where they rest in sites at the margin of sun and shade. As ectotherms, they move into sunlight or retreat to shade to maintain a narrow range of body temperatures. Herein I report on use of anthropogenic features, especially highway guardrails, used by *A. neotesselatus* as habitat.**

Key Words.—*Aspidoscelis sexlineatus*; structures; Colorado; fences; lizards; *Sceloporus consobrinus*.

The Colorado Checkered Whiptail (*Aspidoscelis neotesselatus*) is a triploid, all-female parthenogenetic lizard with a native range endemic to Colorado (Livo 2009). In southeastern Colorado, it occurs in eight counties along the Arkansas River and its tributaries (Walker et al. 2025). Introduced arrays are present in northern Colorado in Adams, Denver, and Douglas counties (Taylor et al. 2015; Livo et al. 2019, 2023). In Otero and Las Animas counties, this lizard is sympatric with its maternal progenitor, the diploid parthenogenetic Common Checkered Whiptail (*Aspidoscelis tesselatus*), and is also broadly sympatric with its diploid gonochoristic paternal progenitor Six-lined Racerunner (*Aspidoscelis sexlineatus*; Walker et al. 1997a). Hybridization of a female *A. tesselatus*, which contributed its diploid genome, and a male *A. sexlineatus*, which contributed a haploid genome, resulted in the formation of the triploid *A. neotesselatus* (Walker et al. 1997a). The dorsal pattern of stripes, bars, and spots along with the streamlined body shape and long tail of *Aspidoscelis neotesselatus* make it easy to distinguish from most other lizards with which it is sympatric in Colorado. The exception to this is distinguishing it from *Aspidoscelis tesselatus* in the limited areas of sympatry. The presence of an irregular white strip on the rear of one or both thighs usually is present on *Aspidoscelis neotesselatus*, while *A. tesselatus* has thighs that lack this stripe and have more prominent spots (Walker et al. 1997a; Livo 2009).

Parthenogenetic *Aspidoscelis* species have often been termed weeds based on their occurrence in disturbed habitats, often in habitats minimally available to sexual species in the genus (Wright and Lowe 1968). *Aspidoscelis tesselatus* and its derived parthenogen *A. neotesselatus* occur in rocky habitats compared to the flatter habitats preferred by the bisexual *A. sexlineatus* (Wright and Lowe 1968). In Colorado, *A. neotesselatus* frequents such habitats as rocky canyons, slopes above drainages (with or without permanent water), and sparsely vegetated hillsides (Walker et al. 1997b; Livo 2009).

Sloped terrain usually occurs in the vicinity of anthropogenic features such as guardrails and bridges of roadways. In many respects, *A. neotesselatus* could be expected at least occasionally to be present in the vicinity of these features simply because of the association of this lizard with slopes. There is no information in the literature pertaining to usage of features such as guardrails and metal chain link fences by *A. neotesselatus*. In a search of volumes 1–54 of Herpetological Review for the terms guardrail and guard rail, I was unable to find any instances of use of these features by whiptails or any other lizard. Regarding other *Aspidoscelis* species, James W. Walker brought to my attention an unpublished manuscript in which he and J.E. Cordes observed the frequent exploitation of guardrails by *Aspidoscelis tesselatus* and the Texas Spotted Whiptail (*Aspidoscelis gularis gularis*) in part of the Palo Duro Canyon system along Highway 207 in Armstrong County, Texas.

Here I summarize the geographically widespread usage of guardrails, bridges, fences, and other selected anthropogenic features by *A. neotesselatus* in Colorado. During my fieldwork between 2018 and 2024, I opportunistically encountered *A. neotesselatus* in association with guardrails and other anthropogenic structures. I attempted to photograph these instances and included this information in my field notes. I have omitted countless occasions where *A. neotesselatus* used human-made or modified slopes associated with bridges, railroad embankments, retaining walls, etc., if they lacked shade-producing structures such as guardrails or chain link fences. I made no effort to apportion observations equally between structure types. In particular, I more frequently observed lizard use of guardrails in southeastern Colorado and fences in northern Colorado. This was due in large part to more general explorations in southeastern Colorado where I checked bridges and other

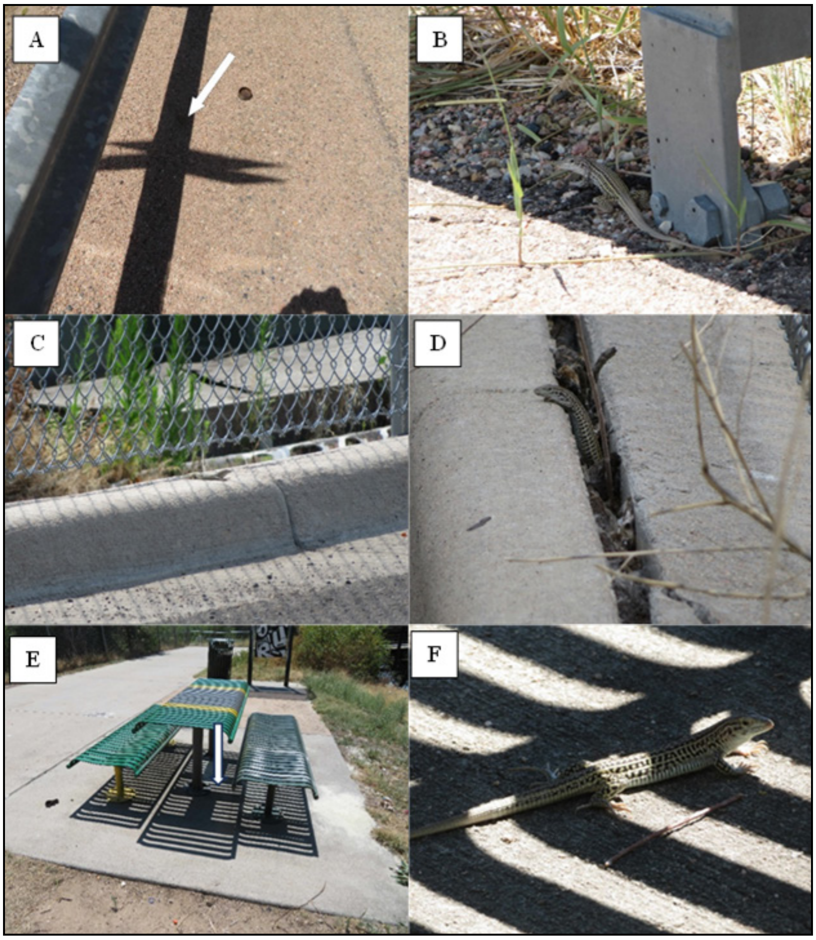

Figure 1. (A) A Colorado Checkered Whiptail (*Aspidoscelis neotesselatus*) in the shade of the guardrail at arrow (Pueblo County); (B) *Aspidoscelis neotesselatus* in the shade of a guardrail near a bridge (Crowley County); (C) *Aspidoscelis neotesselatus* basking on the curb under a chain link fence (Denver County); (D) View from the side of an *Aspidoscelis neotesselatus* in the gap under a chain link fence; (E) Metal table and nearby garbage can used by *Aspidoscelis neotesselatus* as shade (Adams County); (F) Closeup view of *Aspidoscelis neotesselatus* shaded under metal table (Adams County). (Photographed by Lauren J. Livo)

areas along highways with guardrails versus my regular survey efforts in northern Colorado that mostly occurred along a pedestrian/bicycle path where particular fences were present and guardrails absent (Livo et al. 2019; Livo et al. 2022; Walker et al. 2025).

While I observed *A. neotesselatus* both foraging (actively moving, flicking the tongue, and digging) along features such as guardrails and fences (Fig. 1), I more frequently saw them apparently resting either in the shade or at the boundary between sunlit and shaded substrate when associated with those features. The most frequently observed use of anthropogenic features was comprised of guardrails, especially those associated with bridges. Substrates under guardrails included both paved surfaces and gravel/dirt surfaces, depending on location. I observed 38 *A. neotesselatus* using eight different sections of guardrail in five different Colorado counties in both the native range and an area to which this species has been introduced. When I observed lizards at guardrails on one occasion, a repeated visit by me even years apart under appropriate weather and time conditions frequently resulted in additional observations of *A. neotesselatus* at or near the guardrails. Most guardrails were relatively short (< 40 m in length) and I usually saw only one to three lizards along these guardrails on any particular visit to a site, although lizards might also be observed nearby but not using the guardrails. In contrast, I counted eight *A. neotesselatus* along guardrails approximately 1-km long set along a highway as it descended a hillside.

Of 20 observations of *A. neotesselatus* using fencing, 17 took place along a single chain link fence bordering the parking lot near Carpio Sanguinette Park, Denver County (Fig. 1). In four cases between 2018 and 2024, I observed two lizards using this fence at the same time. The lizards basked on the cement curb in the shade of the chain link fence or associated poles or shelter in a gap between the curb and adjacent cement. In addition, these lizards used the shade/sun boundary of fence lines, free-standing poles, and even a metal table and garbage can located on a cement pad (Fig. 1; Appendix Table). In Colorado, I most frequently observed *A. neotesselatus* at guardrails and other anthropogenic features in late morning and early afternoon, after their primary foraging period would be expected to be completed.

Aspidoscelis sexlineatus ranges across eastern Colorado and occurs in sympatry with *A. neotesselatus* at many localities (Walker 1997b; Walker et al. 2025). Despite having observed hundreds of *A. sexlineatus* during my fieldwork, I only have two records where I noted their presence along the shade of a fence (22 July 2018 and 1 June 2019) despite dozens of observations of the sympatric *A. neotesselatus* using these structures. I have a single observation of an *A. sexlineatus* in the vicinity of a guardrail (1 July 2021) in southeastern Colorado, and for this observation the lizard was moving parallel to the guardrail, but a meter or so away where the guardrail appeared to have little if any influence on the lizard.

Use of anthropogenic features is in part due to how these features serve as alternates to natural features used during portions of the daily activity pattern of *A. neotesselatus*. After emerging from overnight shelters, *A. neotesselatus* begin actively foraging. They are most active between about 0800 and 1100 (Aubrey et al. 2019). After foraging, usually by late morning or early afternoon, *A. neotesselatus* often can be observed resting along the margins of vegetation, such as grasses, forbs, and shrubs, in the shaded areas of south-facing boulders, or at the juncture of a ring of grasses or forbes adjacent to bare ground surrounding anthills (pers. obs.) These sites are also used occasionally by the sympatric *A. sexlineatus*. One or more individual *A. neotesselatus* might share these sites, apparently without conflict (Fig. 2). All of these situations allowed the lizards to have their bodies in either sun, shade, or a combination, presumably to maintain a preferred body temperature. The guardrails used by *A, neotesselatus* were along highways and roads with relatively low traffic volume; single-date monitoring occurred at six stations near guardrails with traffic volumes ranging from 335–1,427 vehicles per day (https://dtdapps.coloradodot.info/otis). Although it is possible that *A. neotesselatus* in the vicinity of guardrails sometimes fall victim to vehicular traffic, I did not observe any whiptail carcasses on the roads.

Other lizards make frequent use of anthropogenic habitats. For example, in the neotropics many nocturnal gecko species make extensive use of buildings and walls, especially in the vicinity of lights that attract potential prey items (pers. obs.). Members of the genus *Sceloporus*, which are diurnal lizards, frequently employ anthropogenic habitats as perches and shelters. Indeed, I often observed perching by a saxicolous morph of the Prairie Lizard (*S. consobrinus*) on some of the guardrails at bridges simultaneous with the guardrail use by *A. neotesselatus* including in Crowley, Otero, and Pueblo counties. While *S. consobrinus* occupying guardrails frequently perched on the top of the structures, as they would on boulders or rocky outcrop areas, they also made use of shaded areas at times, either retreating into crevices within the structure or clinging to the shaded faces of upright wooden posts. I never saw the *A. neotesselatus* climb any of the guardrail structures other than low curbs, so the perching *S. consobrinus* and ground-dwelling *A. neotesselatus* effectively used distinctly different parts of the available guardrail habitat.

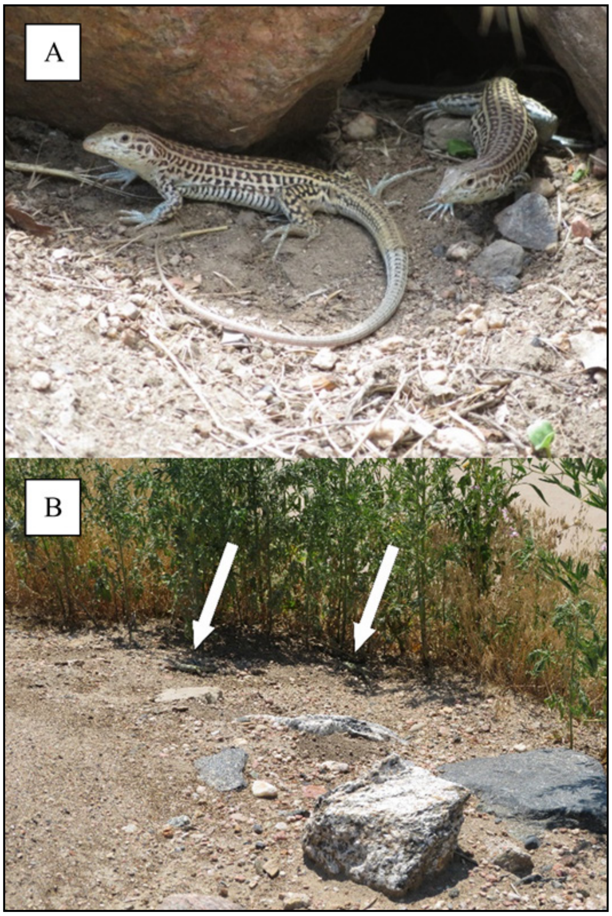

Figure 2. (A) Two Colorado Checkered Whiptails (*Aspidoscelis neotesselatus*) sharing the shade of a rock wall under a bridge in early afternoon (Denver County). (B) Two Colorado Checkered Whiptails (*Aspidoscelis neotesselatus*), indicated by arrows, at margins of opening around an ant hill (Denver County). (Photographed by Lauren J. Livo)

Because vegetation was usually some distance away from the guardrails, grasses, forbs, and shrubs did not provide the usual retreat from potential danger that was typical of natural sites, such as the margins of anthills or other patches of open ground with adjacent vegetation. Although hatchlings were sometimes observed at sites with guardrails, I did not observe any hatchlings using these structures. This may be due to different foraging or thermoregulation needs compared to adult lizards or simply due to the opportunistic character of my observations. With the way the adult lizards oriented themselves, the primary use of these anthropogenic structures appears to be associated with opportunities for thermoregulation.

Acknowledgments.—I thank James W. Walker for encouraging this note and sharing information about related observations of guardrails as habitat for whiptails in Texas. Harry L. Taylor, Todd Wilcox, and Steve Wilcox participated in portions of the fieldwork. Some fieldwork

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Appendix Table. Dates and numbers (in parentheses) of Colorado Checkered Whiptails (*Aspidoscelis neotesselatus*) associated with anthropogenic structures. At several localities, the number of lizards does not reflect the number that may have been in the general area on a particular date but that were observed some distance from the structure.

Guardrails
Adams County: York Street: 30 July 2022 (1). **Crowley County:** Hwy 167: 19 May 2018 (1); 28 July 2018 (1). Hwy 207: 20 May 2018 (2); 2 June 2018 (1); 12 June 2019 (1); 16 May 2020 (1). **Fremont County:** Hwy 210: 1 August 2022 (1). **Otero County:** Hwy 109: 24 June 2020 (3); 25 June 2020 (1). **Pueblo County:** Beulah Road: 17 May 2020 (2); 17 May 2020 (3); 18 May 2020 (8). Hwy 209: 19 May 2018 (2); 2 June 2018 (1); 28 July 2018 (1); 24 August 2018 (2). Burnt Mill Road: 25 June 2019 (2); 12 July 2019 (3).

Fencelines
Adams County: Fence near pedestrian bridge: 5 August 2024 (1). **Denver County:** Carpio Sanguinette Park: 25 May 2018 (1); 28 May 2018 (2); 15 June 2018 (1); 30 June 2018 (1); 27 July 2018 (2); 29 July 2018 (1); 1 August 2019 (1); 7 August 2019 (1); 12 August 2019 (1); 14 August 2019 (2); 1 August 2020 (2); 22 June 2021 (1); 17 July 2021 (1); 19 July 2021 (1); 5 August 2021 (1); 28 August 2021 (1); 30 July 2022 (1). **Pueblo County:** Metal sports field perimeter: 7 July 2018 (4); 24 August 2018 (1).

Metal table/can
Adams County: 19 August 2023 (1); 22 August 2023 (1); 5 August 2024 (1).

Lauren J. Livo for several years conducted research on the Boreal Toad (*Anaxyrus boreas*) in collaboration with the Colorado Division of Wildlife. After receiving a Ph.D. from the University of Colorado, Boulder, USA, she continued her work on the Boreal Toad as a Post-Doctoral Fellow at the University of Colorado. Subsequent to retiring, she has been documenting the geographic distribution and phenology of amphibian and reptile species in Colorado, especially that of various introduced species including the Pond Slider (*Trachemys scripta*), Colorado Checkered Whiptail (*Aspidoscelis neotesselatus*), and Chihuahuan Spotted Whiptail (*A. exsanguis*). (Photograph by Steve Wilcox).

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NOTES

RECORD MAXIMUM SNOUT-VENT LENGTHS OF THE ENDANGERED BLUNT-NOSED LEOPARD LIZARD (*GAMBELIA SILA*)

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***Abstract*.—Large size of individuals in an animal population can confer selective advantages over smaller members. The Blunt-nosed Leopard Lizard (*Gambelia sila*) is an endangered species of the San Joaquin Desert for which the maximum snout-vent length (SVL) of lizards in the 1960s was reported to be 123 mm. Since then, population studies have not reported lizards greater than this maximum size. Here we report record maximum SVL of males in two populations.**

Key Words.—California; lizards; mass; San Joaquin Desert; snout-vent length; SVL.

The size of animals can be an important factor affecting survival and reproduction. For many vertebrates, larger females produce more or larger offspring than smaller individuals (in den Bosch et al. 1998; Germano and Williams 2005; Fokidis et al. 2007; Goncalves et al. 2011). Body size of males can also lead to selective advantages. Large size can benefit territorial males by conferring an advantage defending their territories against other males, which can lead to larger territories and therefore access to more females (Fox et al. 1983; Shine et al. 2000; Candolin and Voigt 2001). In some lizards, larger males have significantly greater bite force in their jaws than smaller females (Verwaijen et al. 2002; Brecko et al. 2008), and bite force was greater for lizards with larger body size than smaller lizards (Donihue et al. 2016). Differences in bite force could allow larger individuals to access more and a wider variety of resources than smaller individuals.

Blunt-nosed Leopard Lizards (*Gambelia sila*) are predators, mainly on arthropods and occasionally small lizards (Montanucci 1965; Tollestrup 1979; Germano et al. 2007). Male *G. sila* are territorial (Montanucci 1965; Tollestrup 1983) and a large male could potentially defend their territory better against smaller males because of its larger size, thereby having greater reproductive success. Larger size, particularly a larger head, may allow *G. sila* a competitive advantage over smaller individuals because of the ability to seize and swallow larger prey. Although *G. sila* has been studied for over 60 y (Montanucci 1965, 1967), the maximum size of 123 mm snout-vent length (SVL) for adult lizards has not been reported to differ from these earliest publications. Here we report record SVLs for male *G. sila* from two populations in the San Joaquin Desert. We use SVL as our measure of size because weight in these lizards and most small animals, especially reptiles, can fluctuate depending on size and type of their last prey, when they last defecated, or if females are gravid.

From 2015 to 2017 we intensively surveyed three sites in the San Joaquin Desert for *G. sila* during the adult and hatchling seasons (April to October) by completing walking surveys on permanent grids approximately two days a week to study demographics and population size. Grids were 300 × 300 m and were either previously established or newly created at Pixley National Wildlife Refuge (Pixley NWR; Tulare County, California), Northern Semitropic Ridge Ecological Reserve (hereafter called Semitropic Ecological Reserve; Kern County, California), and Lokern Ecological Reserve (Kern County, California). From 2018–2024, we scaled back our efforts and conducted 10-d censuses during both the adult (April-July) and hatchling seasons (August-October) on the grids at Semitropic and Lokern Ecological Reserves. At Pixley NWR, 10-d censuses were only completed in 2019 and 2022. At Pixley in 2021 and 2024, only three survey days were completed. No surveys were completed at Pixley in 2023. These demographic censuses were supplemented by a radio-telemetry study at the Lokern site and a nearby oil field study site in 2015 and 2016 (Germano et al. 2024), and additional work at Semitropic Ecological Reserve and Pixley NWR during the same period (unpubl. data). For the demographic study (unpubl. data), we and personnel from the Central Region Lands Unit of the California Department of Fish and Wildlife and the Kern and Pixley NWR of the U.S. Fish and Wildlife Service caught *G. sila* using a modified fishing pole with a loop at the end made from dental floss (Stebbins 1954). We permanently marked each individual adult with a Passive Integrated Transponder (PIT) subdermally or intra-abdominally (Germano and Williams 1993). We collected demographic data, which included measuring their mass (± 0.5 g) and snout-vent length (SVL; ± 1 mm). In line with previous studies by the second author (Germano and Williams 2005; Germano et al. 2024; Germano 2025), SVL was precisely measured using a clear plastic millimeter ruler

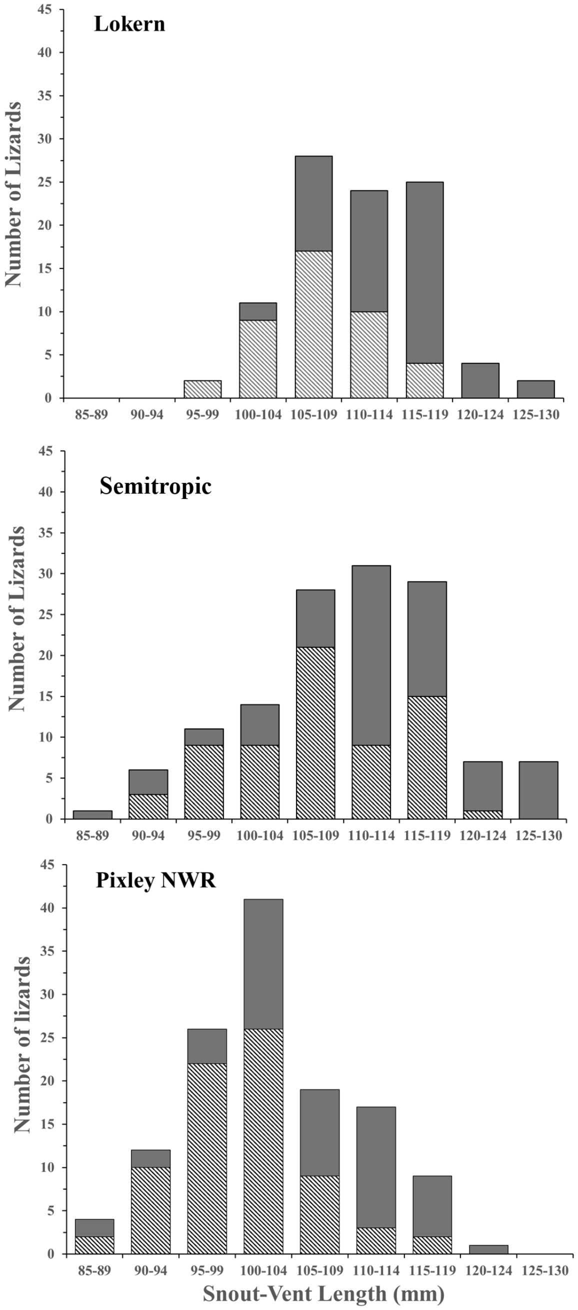

Figure 1. Distribution of the longest snout-vent lengths (mm) of female (light grey) and male (dark grey) Blunt-nosed Leopard Lizards (*Gambelia sila*) caught 2015–2024 at Lokern and Semitropic Ecological Reserves in Kern County, California, and at Pixley National Wildlife Refuge (NWR) in Tulare County, California.

in which the lizard was held against and its body gently stretched to eliminate the lizard from bending its body, thereby shortening its overall length. Our experience suggested that by not straightening lizards, the SVL can be reduced by several millimeters. Mass was recorded using spring scales and taken to either a whole number or to 0.5 decimal when the mark was approximately halfway between whole values. We used the largest size an individual attained, if that individual was recaptured over the study period, to determine the distribution of adult SVLs for each site and the maximum SVLs and mass at each study site.

Of the 54 individual adult male *G. sila* caught 2015–2024 on the Lokern Ecological Reserve, the maximum SVLs measured varied from 102–127 mm (Fig. 1). At Semitropic Ecological Reserve, the SVLs of 67 adult males varied from 89–128 mm, and for 55 males at Pixley NWR, SVLs varied from 86–121 mm (Fig. 1). Based on past studies that reported measurements of *G. sila* (Table 1), the maximum SVLs of the largest males we caught at both the Lokern and Semitropic Ecological Reserves are larger than any previously published data. Prior to our study, the largest male SVL reported in the literature was 123 mm (Montanucci 1965, 1967), and a recent study from Germano (2025) did not change this. The 123 mm SVL male Germano (2025) caught was found during the Lokern grazing study (Germano et al. 2012) on the Lokern Triangle (northwest of Hwy 58 and East of Hwy 33) and weighed 54.9 g, but the largest weight for a male at this site was 63 g for a 116 mm SVL individual (Table 1). The largest SVL of *G. sila* reported by Montanucci (1965) was 123 mm, but he did not give the sex of the lizard obtaining this size. Likely this size was for a male. Both Jennings (1995) and McGuire (1996) also report the largest SVL of *G. sila* as 123 mm, but no attribution is given, and this seems to be taken from the Montanucci (1965) dataset. Using 123 mm SVL as the benchmark, at the Lokern Ecological Reserve, we found two males ≥ 123 mm SVL: one male 125 mm SVL (58 g) and one male 127 mm SVL (51 g). At the Semitropic Ecological Reserve, we found 10 males ≥ 123 mm SVL; two 123 mm (45, 58.5 g); one 124 mm (44 g); three 126 mm (50, 51, 53 g); one 127 mm (56 g); and three 128 mm (48, 51, 59 g). The largest male found at Pixley NWR was 121 mm SVL and it was also the heaviest male at that site weighing 55 g (Table 1). In a recent study on the Elkhorn Plain by Weaver et al. (2024), the greatest SVL for a male *G. sila* was 122 mm and the heaviest male was 56 g (Table 1). Previous demographic work by Germano and Williams (2005) on the Elkhorn Plain found the greatest male SVL to be 118 mm, and the heaviest male was 60 g (Table 1).

Of the 42 individual female *G. sila* we caught 2015–2024 on the Lokern Ecological Reserve (Fig. 1), the largest SVL was from a gravid female that measured 118 mm and weighed 42.5 g. The heaviest non-gravid females at the Lokern Ecological Reserve were two lizards at 37 g (one 113 mm SVL and one 105 mm SVL; Table 1). At the Semitropic Ecological Reserve, we caught 67 individual female *G. sila* 2015–2024, and three females were 118 mm SVL (one non-gravid female weighed 45 g), one was 119 mm SVL (non-gravid: 38 g), and one gravid at 122 mm SVL. The heaviest non-gravid female was the 45 g, 118 mm SVL individual reported above (Table 1). From 2015–2024 we caught 75 individual female *G. sila* at Pixley NWR, and the largest was 118 mm SVL (Fig. 1), was non-gravid, and weighed 33 g.

Table 1. Comparison of the longest snout-vent lengths (SVL) and greatest mass (g) by sex of Blunt-nosed Leopard Lizards (*Gambelia sila*) reported in published studies from the San Joaquin Desert. Montanucci (1965) did not give sizes by sex and only stated longest adult SVL (123 mm). Weaver et al. (2024) and Germano and Williams (2005) worked on the Elkhorn Plain. Weaver et al. (2024) did not differentiate in mass data whether females were gravid, so we excluded female mass data from the table. Data for Germano (2025) were gathered during a livestock grazing study (Germano et al. 2012) at the Lokern Natural Area. Mass values for the longest females are for those not carrying eggs. The longest SVL and greatest mass were not for the same animal, except those marked with an asterisk (*).

	Females		Males	
Reference	SVL (mm)	Mass (g)	SVL (mm)	Mass (g)
Tollestrup (1982)	111	—	120	—
Germano and Williams (2005)	116	47	118	60
Weaver et al. (2024)	114	—	122	56
Germano (2025)	121	58	123	63
This Study - Lokern	118	37	127	58
This Study - Semitropic	122	45	128*	59*
This Study - Pixley	118	36	121*	55*

The heaviest females at Pixley were three females at 36 g (103 mm, 104 mm, 104 mm SVL). Of published records for female *G. sila* (Table 1), the largest SVL at Pixley NWR was 111 mm (Tollestrup 1982). For the Elkhorn Plain, the largest SVL for females from two studies were 114 mm (Weaver et al. 2024) and 116 mm (Germano and Williams 2005). Tollestrup (1982) did not give mass values, but on the Elkhorn Plain, the heaviest non-gravid female was 47 g (Germano and Williams 2005; Table 1). Weaver et al. (2024) did not differentiate whether females were gravid, so we excluded mass values from that study. The largest female recorded on the Lokern grazing study (Germano 2025; at the Lokern triangle mentioned previously) was 121 mm SVL weighing 44.7 g, but the heaviest non-gravid female was 58.1 g with a SVL of 116 mm (Table 1).

We found the largest male *G. sila* at our Lokern and Semitropic Ecological Reserves to be 4–5 mm larger, respectively, than the largest size (123 mm) previously reported by Montanucci (1965) and Germano (2025). Two males at Lokern Ecological Reserve and eight males at Semitropic Ecological Reserve were > 123 mm SVL (Fig. 1). We also found larger females at all sites than past studies (up to this year) that specifically reported female SVLs. Germano (2025) found the largest female at a different part of the Lokern area with an SVL of 121 mm, which is only 1 mm shorter than the largest female we found at Semitropic (Table 1). Weights of both male and female *G. sila* at our survey sites, however, were not unusually large compared to previous reported weights.

We do not know if environmental conditions have changed in the range of *G. sila* that has led to longer individuals or if the method of measuring SVL has contributed to this difference. We are confident that our method of measuring individuals against a clear plastic ruler, while making sure the body is straight, is an accurate method for measuring SVL in lizards. We also know that the distribution of large lizards at Lokern and Semitropic were not specific to a certain field technician or year but distributed throughout the study period. We believe there may be environmental conditions at Lokern and Semitropic that allowed lizards to grow larger than at other sites, but exactly what these conditions might be are unknown. Understanding demographic characteristics of *G. sila* populations throughout their range could help to develop better conservation measures for the species. This may be especially useful for translocation/reintroduction plans, as it is typically recommended to keep similar population groups together.

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NOTES

BLACK SKIMMER (*RYNCHOPS NIGER*) NESTING SUCCESS TRENDS ALONG THE EASTERN SHORE OF THE SAN FRANCISCO BAY, CALIFORNIA

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Abstract.—**The Black Skimmer (*Rynchops niger*), a distinctive colonial waterbird species, is listed as a California Species of Special Concern. Factors affecting their breeding population include limited suitable open nesting habitat, human disturbance, varied food availability, predation (feral animals and gulls), extreme weather, and environmental pollutants. Typically, islet-breeding skimmers are in close proximity to nesting tern species, which provide early warning and defensive behaviors against intruders. Since 2001, the East Bay Regional Park District, California, has been working to establish and enhance a California Least Tern (*Sternula antillarum browni*) colony at Hayward Regional Shoreline located along the eastern shore of San Francisco Bay. In the efforts to assist in the recovery of this state and federally listed endangered species, it resulted in the attraction of breeding Black Skimmers to the site in 2015. For a total of eight breeding seasons between 2015 and 2024, the Black Skimmer has nested successfully, with 90% of nests hatching at least one egg; fledging success was 1.56 fledglings produced per breeding pair. During this same 10-year period, the endangered California Least Tern, and the threatened Western Snowy Plover (*Anarhynchus nivosus nivosus*), American Avocet (*Recurvirostra americana*), and Black-necked Stilt (*Himantopus mexicanus*) have also nested successfully at this location. The results presented on Black Skimmer breeding chronology, hatching, fledging success, and diet in the northern portion of the species range answers data gaps that may help inform future research, protection, and management measures for this special status bird.**

Key Words.—conservation; management; nest success; protection; fledging success; waterbirds.

INTRODUCTION

The Black Skimmer (*Rynchops niger*) is a distinctive and beautiful coastal colonial waterbird, usually seen gliding gracefully low over the water on elegant wingbeats (Fig. 1), with its iconic longer lower mandible slicing through the wet surface in search of fish. This species has an extensive distribution across coastal areas of North and Central America. Breeding colonies are observed along the North American Atlantic Coast as far north as Massachusetts and as far south as parts of Mexico (Clapp et al. 1983; Gochfeld and Burger 1994). On the Pacific Coast, the species winters from southern California south to Baja California and the Gulf of California, as well as the Pacific Coast of Central and South America (Howell and Webb 1995). In California, they typically occupy breeding colonies from mid-April through September, and occasionally into mid-October, with eggs first laid in early May and their last young fledged by late September (Gochfeld and Burger 1994; Molina 1996).

The Black Skimmer is listed as a California Species of Special Concern (Shuford and Gardali 2008) and was first observed in California in Orange County in 1962 (McCaskie and Suffel 1971). The first breeding records for this species in California occurred at the Salton Sea in 1972 (McCaskie et al. 1974) and in south San Diego Bay, San Diego County, in 1976 (Shuford and Gardali 2008). The initial east San Francisco Bay record of nesting Black Skimmers occurred at the Hayward Regional Shoreline, Alameda County, in 1994 (Bob Richmond et al., unpubl. report). The bulk of the breeding population in California can be found in coastal southern California and the Salton Sea, and while no statewide population surveys have been conducted, it is estimated that the breeding population in California ranges between 1,400 and 1,500 pairs (Shuford and Gardali 2008). Factors affecting breeding skimmers include limited suitable open nesting habitat and its continued loss as a result of erosion or vegetation growth on small islands (Molina 2004),

FIGURE 1. Black Skimmer (*Rynchops niger*) flying with elegant wingbeats at the Hayward Regional Shoreline (Tern Town), San Franciso Bay, California. (Photographed by David Riensche).

Figure 2. Black Skimmer (*Rynchops niger*) nest with chicks at the Hayward Regional Shoreline (Tern Town), San Franciso Bay, California. (Photographed by David Riensche).

human disturbance, varied marine food availability, predation (feral animals and gulls), extreme weather, and environmental pollutants (Coburn et al. 2001). It is projected that by 2050, this species, in the face of a changing climate, will have its habitat reduced by as much as 50% (National Audubon Society 2015). Islet-breeding skimmers typically nest in close proximity to nesting tern species that provide early warning and defense against intruders (Gochfeld and Burger 1994).

Black Skimmers are subject to many of the same conservation challenges faced by shorebirds and near shore seabirds, such as loss and degradation of nesting habitat, sea level rise, nest predation, oil pollution, and human disturbance (Hunter et al. 2001; Kushlan et al. 2002; Evers et al. 2010). The ecology, reproductive biology, and behavior of Black Skimmer populations along the Atlantic Coast and the Gulf of Mexico have been the topic of numerous studies (Erwin 1977 1979; Loftin 1982; Custer and Mitchell 1987; Quinn 1989, 1990; Burger and Gochield 1990). Information regarding the Pacific coast population nesting at the Salton Sea, a large interior saline basin in southern California, has also received research attention since the establishment of breeding in 1972 (McCaskie et al. 1974; Grant and Hogg 1976; Grant 1978; Molina 1996). Shuford and Gardali (2008) made recommendations for additional skimmer studies focusing on diet, foraging, provisioning behavior, nest attendance, reproductive success, fledgling success, juvenile survival, adult longevity, recruitment, and the degree of metapopulation mixing among breeders in California. Our research on Black Skimmers in the northern portion of their range in the San Francisco Bay, where there has been limited published records, begins to address some of those recommendations. In an effort to inform regional waterbird monitoring efforts and develop a site-specific conservation plan, we investigated the population trends, nesting phenology, reproductive success, fledgling success, and diet of a Black Skimmer colony coexisting within a California Least Tern colony on the eastern shoreline of the San Francisco Bay.

METHODS

Study site.—The East Bay Regional Park District manages the nesting habitat of Black Skimmers, California Least Terns, Western Snowy Plovers, and Black Oystercatchers (*Haematopus bachmani*) at the Hayward Regional Shoreline (37°37’47”N 122°8’46”W) located along the eastern shore of San Francisco Bay (Riensche 2007; Riensche et al. 2012a, 2015, 2023). We conducted this study from 2015 to 2024 on Island Five (also known as Tern Town), a 0.24-ha (0.6-ac) island created from dredge materials. Vegetative cover on Tern Town has been managed (with mechanical techniques and herbicide treatments) to 5–15% over the years to minimize non-native vegetation spread and encourage state and federally listed ground-nesting bird species. A 10 × 20 m grid system was established for nest surveys and colony monitoring conducted off the island using binoculars.

Data on nest distribution, chronology of nesting, and reproductive success were collected using the Type 1 method (active monitoring inside colony by permitted biologists; Riensche 2007). In this method, monitors entered the colony to mark nests and record the number of eggs and chicks (Fig. 2). This type of intensive monitoring, conducted twice a week (0700–

1700), yields data on clutch size, hatching success, and evidence of predation. Each nest was monitored from when the first egg was laid and continued until all nest fates were determined. During each visit, the number of eggs within each nest was recorded. We assigned nest fates similar to those described in Brooks et al. (2014). The fate of each nest was defined as successful if ≥ 1 egg hatched; this was determined if recently hatched, relatively immobile chicks were observed lying within the monitored scrapes and there was a corresponding reduction in clutch size, or if a sequential decrease in the number of eggs at nests that contained a pipping egg on the previous visit was observed. Nests were recorded as failed if the nest was abandoned (i.e., eggs were cold and/or moisture was seen on the eggshell) or depredated (signs of predation such as broken eggshells and yolk stains and/or evidence of predator tracks). We used Linear Regression to look for trends in number of nests, and we used Stata 14.2 (Stata Statistical Software, College Station, Texas, USA) for statistical analyses.

Diet data were obtained by collecting fish dropped in the colony during the 2020 breeding season. Once collected, the specimens were stored in plastic bags labeled with the collection date. Next, we soaked items in water and cleaned them with a fine artist's paintbrush and then dried in a laboratory convection oven at 150° C for one hour. We gave each specimen a sample number, which was written on the specimen with a fine tip marker. We recorded the following for each specimen: (1) species or lowest taxonomic group possible; (2) total length (in mm) from the tip of the snout to the end of the caudal fin; (3) standard length (in mm) from the tip of the snout to the end of the hypural bone; (4) body depth (in mm), which was the widest part of the fish; and (5) dry weight (g; Riensche et al. 2018). We report standard length results but note that total length measurements can be inaccurate due to caudal fins being broken or chipped on dropped fish.

RESULTS

Breeding Black Skimmers were typically on-site from the end of April through mid-September (2015 to 2024), with the exceptions of 2018 and 2021, when they did not nest (Fig. 3). We and volunteers monitored 78 nests during 403 observation intervals (i.e., one interval is the time period between nest visits) at the colony from 2015 to 2024. Observation intervals ranged from 1–5 d, although 85% of the intervals were 2–3 d. Complete nest clutches with three to four eggs (90%) were more common than clutches with one or two eggs (10%) at this location.

Of the 78 Black Skimmer nests monitored, 90% of the nests were successful. Only eight of the nests failed to produce at least one fledgling, primarily from predation, although abandonment, failure of eggs to hatch, and deterioration of the nesting substrate/scrape all contributed to nest failures as well. In most years, we observed nest initiations in May and first nests hatching in May or June (Fig. 3). Although egg-laying generally started by mid-May and continued into August, there was a well-defined peak of nest hatching from mid-June through July. Due to consistent adult pair counts, which helped to determine nesting locations, we did not detect any renesting attempts by adults at this location. Fledglings first appeared in July and continued into mid-September. Black Skimmer pairs at this site have produced a total of 220 chicks and 122 fledglings (Fig. 4), yielding a mean of 1.56 fledglings/pair. Over the last decade, the number of Black Skimmer nests have increased, although the positive trend is not significant ($F1,8 = 1.69$, $P = 0.229$; Fig. 5). We and volunteers collected 31 dropped fish at the nesting area, belonging to four fish families: Atherinopsidae,

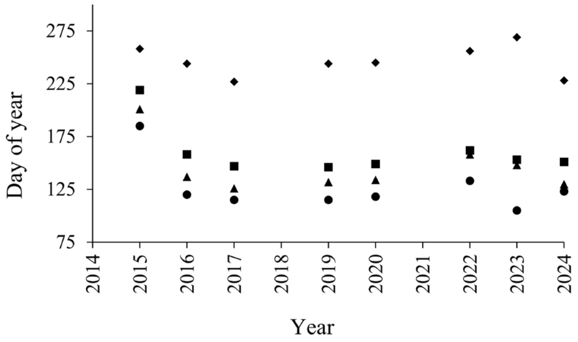

FIGURE 3. Black Skimmer (*Rynchops niger*) nesting chronology at the Hayward Regional Shoreline (Tern Town), San Francisco Bay, California, 2015–2024. Symbols are date first observed (circles), nest initiation date (triangles), date first nest hatched (squares), and date last observed (diamonds). Skimmers are typically at this site from the end of April through the end of August; for reference, Day of Year tick marks for non-leap years on y-axis ranges are 75 (March 16), 125 (May 5), 225 (August 13), and 275 (October 2). The skimmers did not nest at the site in 2018 and 2021.

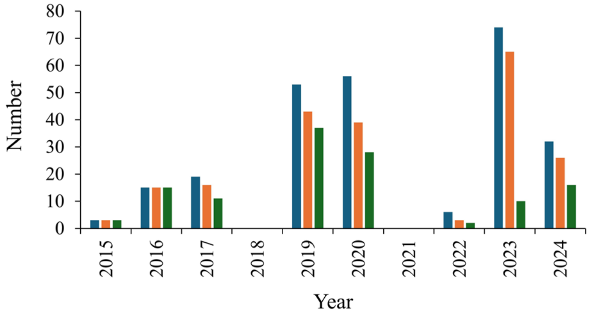

FIGURE 4. Black Skimmer (*Rynchops niger*) breeding success at the Hayward Regional Shoreline (Tern Town), San Francisco Bay, California from 2015 to 2024. Bars shown are numbers of eggs (blue), chicks (orange), and fledglings (green). The average nest success was 90% and fledgling per pair ratio was 1.56.

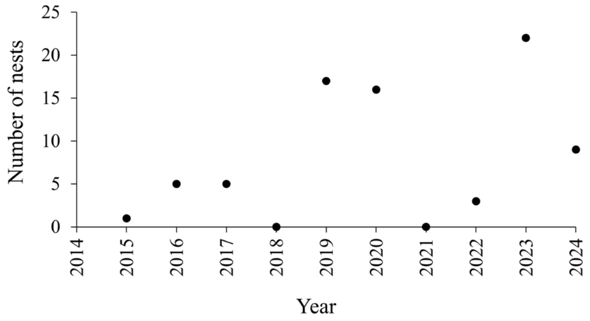

Figure 5. Black Skimmer (*Rynchops niger*) nesting trends (2015 to 2024) at Tern Town (Hayward Regional Shoreline).

Gobiidae, Cottidae, and Gasterosteidae (Table 1). Most dropped prey were atherinopsids, including Jacksmelt (*Atherinopsis californiensis*), California Grunion (*Leuresthes tenuis*), Topsmelt (*Atherinops affinis*), and Mississippi Silverside (*Menidia audens*). The mean standard length of the dropped fish was 66 mm (range of values, 33–121 mm; Table 1).

Discussion

While some natural bird populations are suffering declines, the California Black Skimmer population is increasing in numbers since the first observation in 1962 (Small 1963; Collins and Garret 1996). The first San Francisco Bay nesting record for Black Skimmer occurred in 1994 within Forster's Tern (*Sterna forsteri*) colonies in Santa Clara and Alameda County (Layne et al. 1996). This species continued to nest in small numbers in the San Francisco Bay through 2005 (Rogers et al. 2004). Previously, there were up to two dozen nesting pairs reported at the Redwood Shores in southern San Mateo County (Shuford and Gardali 2008), and surveys in the south San Francisco Bay in May 2019 recorded five Black Skimmers engaging in breeding behaviors and roosting at two island locations (Foster City/Redwood Shores and Moffett; Schacter et al. 2023). With these two exceptions, there have been limited detailed, published studies of the nesting and foraging biology of the Black Skimmer in the region.

Black Skimmers are extremely social, colonial nesting waterbirds requiring bare substrate and isolation from terrestrial predators and other disturbances. Colonies normally are established on small, constructed islands or the remote sections of eroded levees (Shuford and Gardali 2008). They characteristically nest with terns (*Sterna* spp., Gochfeld 1978) and depend on the aggressiveness of terns towards intruders (Gochfeld 1978; Erwin 1979) to thwart potential predators. Their nesting associates include Caspian Tern (*Hydroprogne caspia*), Elegant Tern (*Thalasseus elegans*), Gull-billed Tern (*Gelochelidon nilotica*), Royal Tern (*Thalasseus maximus)*, Black-necked Stilt, and American Avocet (Shuford and Gardali 2008), and in Florida, the Least Tern (*Sterna antillarum*; Kale and Maehr 1989). Tern Town meets some of these important nesting requirements for Black Skimmers by providing a low vegetated area with dredged materials (Loftin and Smith 1996) and several associate species to help defend against predators.

Table 1. Species composition, mean standard lengths (MSL in mm), range of values of standard lengths (RSL in mm), and numbers (n) of dropped prey collected from the Black Skimmer (*Rynchops niger*) nesting area at Tern Town, San Francisco Bay, California, in 2020.

Species	MSL	RSL	n
Atherinopsidae			
California Grunion (*Leuresthes tenuis*)	109.6	96–121	5
Mississippi Silverside (*Menidia audens*)	77		1
Jacksmelt (*Atherinopsis californiensis*)	49.7	38–62	6
Topsmelt (*Atherinops affinis*)	85.5	75–101	4
Unidentified Atherinopsid	33		1
Cottidae			
Unidentified Sculpin	45		1
Gasterosteidae			
Three-spined Stickleback (*Gasterosteus aculeatus*)	43		1
Gobiidae			
Unidentified Goby	55.9	35–81	12
Total	66.4	33–121	31

Adult skimmers attempt to nest every year (Gochfeld and Burger 1994) and will re-nest up to three times within a breeding season if nest failure occurs (Clapp et al. 1983; Gochfeld 1976). Colonies may fail due to predation, overwash during extreme high tides, or other factors, and skimmers may attempt renesting at different sites (Snipes and Sanders 2012). Because nest initiation within a colony is asynchronous, it is recommended that multiple visits are necessary to document maximum colony size (Brooks 2011; Gochfeld 1979). Reports of hatching success among populations of Black Skimmer in North America is highly variable (Gochfeld and Burger 1994). At the extreme southern end of California, in a 3-y study at the Salton Sea, Molina (1996) reported a highly variable hatching success rate for nests of known fate ranging from 27% in 1994 to 71% in 1993. At our location, in the extreme northern limits of the species nesting range in California, we report a 90% nest success rate, ranging from 33% (in 2022) to 100% (in 2015–2017, 2020, 2023). Successful nesting attempts at our study site have produced a total of 220 chicks, resulting in 122 fledglings. The mean 1.56 fledglings per nest obtained along the eastern shore of the San Francisco Bay are higher than those reported for breeding skimmers in Virginia at 0.40 (Erwin 1977), New Jersey at 0.39 (Burger 1982), and South Carolina at 1.2 (Blus and Stafford 1980).

Additionally, Brooks et al. (2014) reported that Black Skimmer nest survival was positively related to clutch size, with the odds of a nest producing at least one fledgling when clutch size was large (≥ 3 eggs) being 2.7 times higher than when a clutch size was small (< 3 eggs). Dinsmore (2008) suggested the effect of clutch size on nest fledging may be related to a combination of female condition or age, location of nest within the colony, and other factors. Experienced breeders appear to have larger clutches than first-time breeders and tend to nest at the center of colonies, where they may be less vulnerable to predation or overwash (Burger and Gochfeld 1990). Clutches with three or four eggs (90%) were more frequent than clutches with one or two eggs (10%) at our location, and this may suggest the Black Skimmers at Tern Town are experienced breeders and/or the habitat is well protected from potential predators (Riensche et al. 2012b).

Skimmers typically forage at dawn and dusk (Wilson 1995) on small fish and possibly crustaceans (Leavitt 1957) by cutting or skimming the surface of the water with their lower mandible in the calm shallows of harbors, lagoons, bays, estuaries, ponds, and river channels (Shuford and Gardali 2008). Periodic monitoring of Black Skimmer diet and egg contents may help identify trends and limiting factors to elucidate possible reasons for reproductive failure (Coburn et al. 2001). Molina (1996) reported that the quantitative information of foraging behavior and diet for the Black Skimmers at the Salton Sea was missing, but that small marine fish like Ronco Croaker (*Baridiella icistia*) and tilapia (*Tilapia* sp.) were occasionally regurgitated by chicks during handling or dropped at the nest. The dropped fish at the Tern Town location is comprised of species common to estuaries and brackish water environments near this nesting site, with slightly over 50% composed of silversides (Atherinopsidae). Wilson (1995) also reported that the most abundant prey items brought to the nesting colony in Bolsa Chica, California, included members of family Atherinidae (Topsmelt, California Grunion, and Jacksmelt), all species we identified in the dropped fish at Tern Town. While we show only one year of dropped fish results here, continued collection of dropped prey may give us better insights into the diet and foraging behaviors of the Black Skimmer colony at Tern Town.

Black Skimmers, like other waterbirds, need access to suitable nesting and brood-rearing habitat (typically an island, or otherwise protected from predators), as well as adjacent foraging and roosting habitat (Erwin et al. 1993; Law and Dickman 1998). Species using coastal habitats may need to adapt and migrate to more suitable habitats to persist in the future (Parmesan 2006; Loarie et al. 2009) due to expected sea level rise. Sea level rise will adversely affect coastal waterbirds through direct mortality of nests and young (Pol et al. 2010), as well as indirectly through the loss of habitat (Hunter et al. 2015). It has been shown that climate change driven population shifts are occurring in New Jersey, where the probability of nesting Black Skimmers coexisting within Least Tern colonies increased significantly over a period of 44 y (Tattoni et al. 2020). Furthermore, Brooks et al. (2014) described that for every 10 cm increase in estimated tide height in South Carolina, the odds of Black Skimmer nesting success decreased by 33%.

The San Francisco Bay has lost much of its historic tidal wetlands, approximately 140 km2 of which were converted to commercial salt production (https://sfestuary.org/wp-content/uploads/2012/12/1Habitat_Goals.pdf). Looking toward future conservation efforts in the region, as Black Skimmers, Least Terns, and Western Snowy Plovers overlap in space and time, any single disturbance now has the potential to negatively impact more individuals and more listed species. Greater effort to understand the limiting factors that drive changes in the distribution of these special status species is critical for informing population recovery efforts in this rapidly changing world. Ultimately, upcoming conservation work focusing on comparing the population dynamics of multi-species colonies within managed ponds, nest and chick survival, and diet information will be needed to develop beneficial management strategies.

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PEER-EDITED NOTE

SOCIAL INTERACTIONS WITHIN AND BETWEEN SPECIES BY LITTLE STRIPED WHIPTAILS (*ASPIDOSCELIS INORNATUS*)

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***Abstract*.— The Little Striped Whiptail (*Aspidoscelis inornatus*) is a bisexual lizard native to the Chihuahuan Desert. Aside from being involved in a few hybridization events, nothing is known about its social behavior or interspecific interactions. At the Dalquest Desert Research Station (Presidio and Brewster counties, Texas, USA), we documented aggressive intraspecific encounters involving postural displays, biting, and pursuit, indicating potential territorial behavior. We also observed interspecific interactions, including avoidance of the Greater Earless Lizard (*Cophosaurus texanus*) and aggressive displacement of the Common Checkered Whiptail (*Aspidoscelis tesselatus*). Our observations provide insight into the social behaviors of *A. inornatus* and underscores the need for further study of its social dynamics.**

Key Words.—aggression; Chihuahuan Desert; lizard; social behavior; Texas.

The Little Striped Whiptail (*Aspidoscelis inornatus*; Fig. 1) is a bisexual species of lizard (Wright and Lowe 1993) found in the Chihuahuan Desert with its range extending from central Mexico into the southwestern U.S. (i.e., western Texas, New Mexico, and Arizona; Duellman and Zweifel 1962; Wright and Lowe 1993; Stebbins 2003). Previous research on *A. inornatus* reported their hybridization with congeners that are normally parthenogenetic: the New Mexico Whiptail (*A. neomexicanus*); the Chihuahuan Spotted Whiptail (*A. exsanguis*); and the Desert Grassland Whiptail (*A. uniparens*; Wright and Lowe 1993), but nothing is known about *A. inornatus* socially, either within or between species.

We observed both intra- and interspecific interactions while conducting focal observations of *A. inornatus* for a separate study at the Dalquest Desert Research Station in Presidio and Brewster counties, Texas, USA (29.563°N, 103.806°W). The habitat where we observed lizards was Chihuahuan Desert scrub, characterized by open patches of sand interspersed with vegetation, primarily Creosote Bush (*Larrea tridentata*), cactuses (e.g., *Cylindropuntia* spp. and *Echinocereus* spp.), monocot shrubs (e.g., Lechuguilla, *Agave lechuguilla*, Green Sotol, *Dasylirion leiophyllum*), and Ocotillo (*Fouquieria splendens*; Fig. 2). We observed one aggressive encounter between two male *A. inornatus* on 17 June 2025, at about 1100 (both with a snout-vent length = 56 mm). Initially, we saw one of our marked *A. inornatus* alone beneath some vegetation. Shortly afterward, a second unmarked *A. inornatus* emerged from a dense patch of vegetation and moved toward the shade of a nearby Creosote Bush located near the first male. Both lizards emerged from cover and approached each other. When the two saw each other, both lizards paused briefly, elevated their postures, and inflated their throats before simultaneously charging. The first lizard initiated physical contact by biting the second on the neck, triggering a vigorous struggle, with both individuals biting and rolling over each other repeatedly. The first lizard eventually bit the snout of the second male (Fig. 3). The second lizard attempted to pull away while the first lizard maintained its grip. The resulting tug-of-war lasted approximately 3 min before the second lizard successfully escaped and fled (> 128 m), passing between observers. The first lizard briefly pursued the second (for about 1 m) before breaking off and returning to the original site of the interaction. Following the aggressive interaction, the first lizard remained in the area for approximately 40 min and revisited the same cluster of vegetation where the aggressive encounter occurred multiple times. The first individual chased another smaller *A. inornatus*, located near the location of the described aggressive encounter, out of the area.

FIGURE 1. Adult male Little Striped Whiptail (*Aspidoscelis inornatus*) on the study site in Texas. (Photographed by Maria Eifler).

Figure 2. The habitat where Little Striped Whiptails (*Aspidoscelis inornatus*) occurred at the Dalquest Desert Research Station, Texas, with the station buildings visible in the background. (Photographed by Jasmyn Zimmerman).

Figure 3. Aggressive interaction involving two male Little Striped Whiptails (*Aspidoscelis inornatus*) on 17 June 2025. The lizard on the left is described in the text as the first lizard, and the one on the right is the second lizard. (Photographed by Maria Eifler).

We also observed interactions between *A. inornatus* and two other lizard species on our study site. On 11 June 2025 at 1643, we observed an interspecific encounter between *A. inornatus* and a Common Checkered Whiptail (*A. tesselatus*; a parthenogenetic, all-female species). The two individuals were similar in size even though *A. tesselatus* can be larger than *A. inornatus* (Stebbins 2003). We speculate that the *A. tesselatus* was probably a young individual whereas the *A. inornatus* was a large adult. While foraging, the *A. tesselatus* entered the shaded area between two bushes, with the *A. inornatus* simultaneously approaching the bushes from a different direction. Upon seeing the *A. tesselatus*, the *A. inornatus* froze momentarily, while the *A. tesselatus* continued foraging, apparently unaware of the *A. inornatus*. When the *A. tesselatus* came within about 30 cm of the *A. inornatus*, the *A. inornatus* elevated its posture and charged forward. The *A. tesselatus* reacted when the *A. inornatus* came within a body length, retreating rapidly and fleeing past the observers. The *A. inornatus* pursued briefly before turning back to the original bush, while the *A. tesselatus* continued moving in the opposite direction, resuming foraging activity.

On 22 June 2025 at 1524 h during a brief interspecific encounter, we saw another *A. inornatus* moving toward a larger Greater Earless Lizard (*Cophosaurus texanus*). Like *A. inornatus*, *C. texanus* are insectivorous (Maury 1995) making them potential competitors. Upon observing the *Cophosaurus*, the *A. inornatus* abruptly darted away.

Our observations provide insight into the behavior of *A. inornatus* during social interactions, conspecifically and with other species. We interpret their behavior as indicating the possibility of home ranges that overlap, with some areas potentially defended (i.e., territoriality) against other *A. inornatus*, as well as other lizard species.

In addition, the possibility of predation occurring between lizard species merits investigation. Further study is needed to understand *A. inornatus* social dynamics, both within and between species.

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NOTES

FLEA SHARING BETWEEN THE SAN JOAQUIN ANTELOPE SQUIRREL (*AMMOSPERMOPHILUS NELSONI*) AND GIANT KANGAROO RAT (*DIPODOMYS INGENS*)

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***Abstract*.—We examined flea sharing between the San Joaquin Antelope Squirrel (*Ammospermophilus nelsoni*) and the Giant Kangaroo Rat (*Dipodomys ingens*), two species that often share burrows and habitat in the San Joaquin Desert of California. We trapped small mammals and collected fleas to identify which species were present on each host. Many flea species that parasitize small mammals are generalist fleas (found on several host species) and live primarily in host burrows where moisture levels are higher than the external environment and temperatures remain lower. Based on these two suppositions, we expect that fleas may be shared between San Joaquin Antelope Squirrels and Giant Kangaroo Rats. We found the fleas *Hoplopsyllus anomalus* and *Thrassis augustsoni* on both species. Our findings indicate that the shared use of burrows likely facilitates the exchange of these ectoparasites.**

Key Words.—ectoparasites; Heteromyidae; San Joaquin Desert; Sciuridae; Siphonaptera.

Since its description and naming in 1916, the San Joaquin Antelope Squirrel (*Ammospermophilus nelsoni*) has been found in parts of Kern, Kings, San Benito, San Luis Obispo, and Fresno counties in the San Joaquin Desert of California (Taylor 1916; Cypher et al. 2021) and has been state-listed as Threatened since 1980 (USFWS 1998). The limited distribution of the species and the encroachment of development and agricultural use in the region have led to a decrease in numbers. *Ammospermophilus nelsoni* prefers desert grassland and shrubland habitats often shared with Giant Kangaroo Rats (*Dipodomys ingens*), and sometimes Heermann's Kangaroo Rats (*D. heermanni*; Best et al. 1990; Williams and Kilburn 1993; Cypher et al. 2021).

The San Joaquin Antelope Squirrel often uses burrows dug by *D. ingens*, thus burrow sharing between the two species may be common and potentially allows for exchange of ectoparasites, such as fleas (Taylor 1916; Best et al. 1990). The diurnal *A. nelsoni* has many burrows in the shared habitat rather than using one home burrow and they move from burrow to burrow while foraging (Cypher et al. 2021). Other researchers describe the nocturnal *D. ingens* as digging burrows and *A. nelsoni* using their abandoned burrows. Hawbecker (1953) and Best et al. (1990) stated that burrows dug by *D. ingens* and *A. nelsoni* are indistinguishable.

Hawbecker (1951) described *D. ingens* as the dominant rodent species in its territory, although other species of small mammals may be present. Tabor et al. (1993) stated that *A. nelsoni* may enter burrows of or with *D. ingens* to rest or to escape the heat. While trapping an area that supports both *D. ingens* and *A. nelsoni*, we found two species of fleas on both rodents.

On 3 and 4 March 2025, and 18–21 March 2025, we conducted small mammal trapping in the Tumey Hills, approximately 80 km west of Fresno, California, and 16 km east of Panoche Valley, California (Township 15S, Range 12E, Section 34; elevation 427 m). We placed 20 Sherman live traps (7.6 × 9.5 × 30.5 cm; H.B. Sherman Traps, Inc., Tallahassee, Florida) and 20 Tomahawk galvanized wire traps (60.1 × 15.2 × 15.2 cm; Tomahawk Live Trap, Hazelhurst, Wisconsin) together along a single 450 m transect with pin flags marking active burrows. As temperatures were cool during the day, we opened squirrel traps from sunrise to sunset, and Sherman traps were open throughout the night. We extracted fleas by hand and then stored the fleas in 70% isopropyl alcohol. We processed fleas using standard methods and identified using multiple references (Hubbard 1947; Augustson 1953; Hopkins and Rothschild 1962; Stark 1970; Campos 1971).

We inspected 11 *Dipodomys ingens* and 19 *Ammospermophilus nelsoni* for ectoparasites but only found fleas on three *D. ingens* and on eight *A. nelsoni* (we did not record the sex of the rodents when fleas were collected; Table 1). We found the fleas *Hoplopsyllus anomalus* and *Thrassis augustsoni* on both rodents, and the flea *Meringis californicus* only on *D. ingens*. Flea collection was opportunistic if the parasites were easy to remove by hand; our collection efforts were expeditious to limit handling time. We do note that we saw *D. ingens* and *A. nelsoni* using the same burrows along the transect.

Egoscue (1985) listed *Thrassis augustsoni* and *Meringis californicus* from San Joaquin Kit Foxes (*Vulpes macrotis mutica*) collected in Kern County, California. He believed these fleas came from *A. nelsoni*, *D. ingens*,

Table 1. Fleas collected from San Joaquin Antelope Squirrel (*Ammospermophilus nelsoni*) and Giant Kangaroo Rat (*Dipodomys ingens*) in Spring 2025 in the San Joaquin Desert, California. Note that while 17 fleas were collected, three were lost when we dropped a vial. The sample size (n) refers to all individuals that we inspected for fleas but not the number of individuals that had fleas.

No. fleas collected	Sex of fleas	Flea species
Ammospermophilus nelsoni (n = 19)		
2	F	*Thrassis augustsoni*
5	M	*Hoplopsyllus anomalus*
4	F	*Hoplopsyllus anomalus*
Dipodomys ingens (n = 11)		
1	M	*Meringis californicus*
1	F	*Hoplopsyllus anomalus*
1	F	*Thrassis augustsoni*

and *D. heermanni*, which kit fox eat. *Ammospermophilus nelsoni* is a known host of several flea species, including *T. augustsoni, H. anomalus*, and *Meringis parkeri*. In the same habitat, *D. ingens* is a host for *M. californicus* and *H. anomalus* (Hubbard 1949; Hawbecker 1953, 1959; Best et al. 1990; Williams and Kilburn 1993).

What are the consequences of this flea sharing? We posit that, as long as flea load per host does not increase significantly, that both species of rodent will transport fleas within the habitat. The bacterium *Yersinia pestis* sometimes is found in fleas and is lethal when transferred to rodents (Butler et al. 1982; Wimsatt and Biggins 2009). If no bacterial pathogen enters the flea populations, host numbers should remain steady, barring unforeseen outside events such as increased habitat degradation. In conclusion, we think that *A. nelsoni* and *D. ingens* share flea species as a result of their use of shared territory and burrows. Future work using data from common shared burrows would supplement our findings.

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ASH BATHING BY WESTERN GRAY SQUIRRELS AND WILD TURKEYS IN AN OAK FOREST OF CALIFORNIA

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Abstract.—**Dust bathing, including in ash, is a common behavior thought to help animals remove ectoparasites like ticks that transmit harmful pathogens. The acaricidal efficacy of bathing in ash depends on the innate properties of ash and willingness of animals to use it. California is ideal for studying this phenomenon given widespread tick infestations on many wildlife species as well as frequent wildfires and controlled burns producing wood ash. We investigated the ash bathing preferences of California wildlife in a predominantly Blue Oak (*Quercus douglasii*) Forest for three common local tree species: Coast Redwood (*Sequoia sempervirens*), California Bay Laurel (*Umbellularia californica*), and Tasmanian Blue Gum Eucalyptus (*Eucalyptus globulus*), which have been evaluated previously for their effects on local tick species. Remote cameras recorded 1,674 independent wildlife visits to piles of ash placed *in situ* over 5 mo. Among the 45 detected ash bathing events, the primary bathers were Western Gray Squirrels (*Sciurus griseus*), which used all three types of ash, and Wild Turkeys (*Meleagris gallopavo*), which used Coast Redwood and Tasmanian Blue Gum Eucalyptus ash. Dark-Eyed Juncos (*Junco hyemalis*), although frequently observed near all ash types, were only observed bathing once in California Bay Laurel. Our findings suggest that some wildlife tick-hosts actively bathe in ash with known acaricidal properties, but we did not detect a significant preference among ash types. Wildfires and controlled burns could contribute to tick control by providing wildlife with natural acaricides. Prioritizing controlled burning of plant species with acaricidal properties could enhance the potential of wildfire management to contribute to tick-borne disease mitigation.**

Key Words.—behavior; dust bathing; ectoparasite; fire; *Meleagris gallopavo*; *Sciurus griseus*; squirrel; tick-borne disease; turkey.

INTRODUCTION

Dust bathing, widely practiced by numerous mammals and birds, is thought to aid in ectoparasite removal, cleaning, thermoregulation, and scent marking (Eisenberg 1963; Branch 1993; Rees 2002; Clayton et al. 2010). Among the many species of wildlife that are known to dust bath, published descriptions of dust bathing in the Western Gray Squirrel (*Sciurus griseus*) are limited, though other squirrel species have been observed using dust baths including the American Red Squirrel (*Tamiasciurus hudsonicus*; Ferron 1976), chipmunks (Sciuridae; Johnston 1998), and several species of ground squirrels (Sciuridae; Steiner 1974; Long and Smith 2023). Wild Turkeys (*Meleagris gallopavo*) also have also been documented dust bathing (Buchholz 1995; Miller 2018). During this process, animals lay down and roll in dust composed of naturally occurring substrates, such as soil, sand, or ash; however, the substrate preferences for dust bathing among wildlife remain largely undocumented, limiting our understanding of how these materials might be used and their potential role in ectoparasite removal.

For domestic animals, ash is specifically provided for dust bathing and used to control ectoparasites (Hakbijl 2002; Banjo et al. 2009; Gabanakgosi et al. 2012; Moyo et al. 2015; Wanzala 2017). Rubino (2024) noted that fine particles of ash may abrade arthropod exoskeletons, block respiratory spiracles, or induce chemical toxicity, ultimately leading to parasite death. The study also experimentally showed that ash from California Bay Laurel (*Umbellularia californica*) and Tasmanian Blue Gum Eucalyptus (*Eucalyptus globulus*) could successfully kill the Western Black-legged Tick (*Ixodes pacificus*), the primary vector of the pathogens that cause Lyme disease (*Borrelia burgdorferi*) and granulocytic anaplasmosis (*Anaplasma phagocytophilum*) in the western U.S. In contrast, ash from Coast Redwood (*Sequoia sempervirens*) was ineffective.

Wildfires and controlled burns are both common in California. These fires can kill ticks outright, reduce habitat for ticks and their hosts, and generate ash that could provide wildlife with a natural acaricide (Gallagher et al. 2022; Brown et al. 2023). Specific ash bathing preferences of wildlife are unknown, however, including whether wildlife seek ash from tree species with higher acaricidal properties. This knowledge could guide prioritization of tree species during controlled burns, highlighting the potential for controlled burns to simultaneously support ecosystem health while aiding in disease control. We aimed to identify patterns in ash bathing among wildlife species and how environmental factors (vegetation composition, canopy cover) influence these patterns. We expected that wildlife, when provided these same three types of ash, would exhibit preferences for more acaricidal ashes, particularly those produced from California Bay Laurel and Tasmanian Blue Gum Eucalyptus over those produced from Coast Redwood.

METHODS

Study site.—We conducted our study at the Quail Ridge Reserve in Napa County, California (38°49′04″N, 122°14′28″W). The reserve supports roughly 18 mammal, 130 bird, and 20 reptile and amphibian species (UC Davis

Natural Reserve System. 2004. Natural History of Quail Ridge. Available from https://naturalreserves.ucdavis.edu/quail-ridge-reserve/natural-history-of-quail-ridge [Accessed 20 January 2025]). The mean temperature in 2024 was 15.6° C, with summer highs frequently exceeding 32.2° C (112 d), and annual precipitation totaling 839 mm (www.ncdc.noaa.gov). We selected four study sites in Decker Canyon (Fig. 1), a dominantly Blue Oak woodland within the most mesophytic habitat of the reserve, influenced by its proximity to a marina and low elevation (UC Davis Natural Reserve System. 2004. *op. cit.*). Scattered California Bay Laurel is found in the Canyon, whereas Tasmanian Blue Gum Eucalyptus and Coast Redwood are absent, no Eucalyptus or Coast Redwood forests occur nearby.

Ash production.—We produced ash off-site in a cleared 100 m^2 area of bare earth using a kiln made from stacked concrete and bricks. We rinsed the kiln between burns to reduce residues from previously burned vegetation. We obtained wood used to produce ash from fallen branches of California Bay Laurel, Tasmanian Blue Gum Eucalyptus, and Coast Redwood within Alameda, Yolo, and Sonoma counties. After burning, we crushed the ash into powder with a rubber mallet to ensure consistency in textural properties.

Experimental design.—Within Decker Canyon, we non-randomly selected four sampling sites (A-D) consisting of 10-m diameter plots containing at least three oak trees for mounting cameras spaced a minimum of 3 m apart and at least three 1 m^2 areas of clear ground. At each site, we used these clear areas to establish ash piles (one for each ash type), so that all three ash types were simultaneously represented at each site. Each site was at least 100 m from the others. We deployed three camera (Hyperfire 2; Reconyx, Holmen, Wisconsin, USA) traps per site 20 June 2024 at Sites A, B, and C, and 2 July 2024 at site D. The cameras were installed on oak trees at least 3 m apart (Fig. 2) to minimize the likelihood of capturing the same animal ash bathing in one ash pile in the adjacent camera. We mounted cameras approximately 0.5 m off the ground using wire and mounting straps on the north side of trees to prevent glare from the rising and setting sun. We angled each camera downward 70° to aim at a single ash pile approximately 2 m from the camera. Ash piles consisted of 470 mL of finely ground ash spread in an approximate diameter of 0.6 m and a height of < 1 cm. We replaced the ash every 12 to 28 d (mean = 17 d), rotating the ash type in front of each camera every 4 weeks by sweeping away old ash residues before depositing a different ash type.

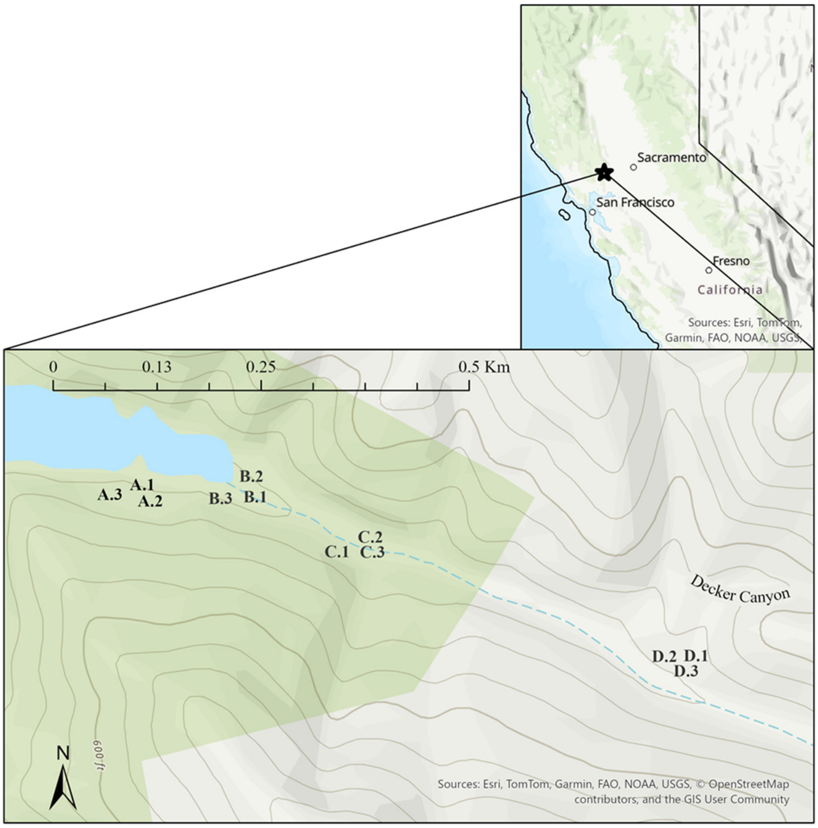

Figure 1. Locations of sites A, B, C, D at Quail Ridge Reserve, California, where camera trapping surveys of wildlife activity were conducted June-October 2024. The grouped letters (e.g. A.1, A.2, A.3) give the locations of mounted cameras. Star in inset map shows reserve location within California.

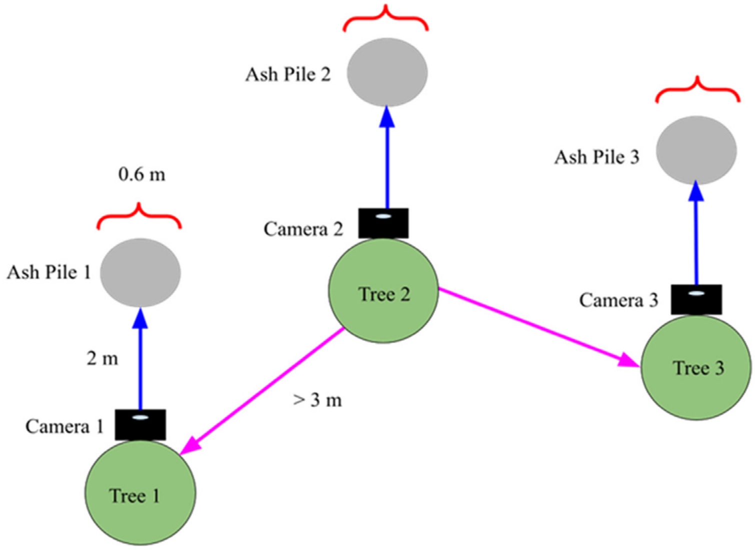

Figure 2. Representative layout of camera-traps deployed at sites A, B, C, and D, at Quail Ridge Reserve, California, for detection of wildlife behaviors at ash piles from June-October 2024.

Trapping methods.—After being deployed in June and July, cameras remained active until 20 October 2024 ahead of rain, which typically starts in the fall in the area we studied (https://www.wunderground.com/history/monthly/us/ca/sacramento/KSMF/date/2024-10). We programmed each camera to take motion-activated pictures instead of video footage due to memory constraints, with detection sensitivity set to high to record animal movement. Each camera took five photographs when triggered, with no delay between triggers. To avoid overexposed photos, we placed green masking tape over half of the LED lights on each camera. Additionally, we trimmed back vegetation within the visual range of each camera to minimize false camera triggers. Minimal trimming was required because our sites had relatively sparse understory vegetation.

Vegetation assessment.—On 19 March 2025, we collected vegetation data at each camera site. We assessed tree density by identifying to species and counting all trees with a diameter > 7.62 cm (3 in) that were within 2 m of each camera trap. We quantified canopy cover from digital photographs taken facing upward 1 m above each ash pile and analyzed using ImageJ software (Schneider et al. 2012). In ImageJ, photographs were first converted to grayscale, then a brightness threshold was applied to distinguish canopy pixels from sky pixels. We calculated the proportion of pixels classified as canopy relative to the total pixels in the image to determine percentage canopy cover (Goodenough and Goodenough 2012). We further recorded key habitat features such as hip level vegetation density, leaf litter abundance, and proximity to roads and water.

Statistical analyses.—For each camera image, we recorded location, date and time, and animal species and behavior including: (1) animals bathing in the ash; (2) animals otherwise interacting with the ash; or (3) moving near the camera without interacting with the ash. We defined ash bathing as rolling, lying prone, or feather-fluffing in the ash (Eisenberg 1963; Joubert 1972; Van Liere and Bokma 1987). Non-bathing behaviors were categorized as: (1) walking on (visible leg movement across consecutive frames); (2) standing on; (3) sitting on; (4) foraging on; or (5) no ash interaction.

We considered visits occurring > 15 min apart from other visits to be independent (Rendall et al. 2014). We excluded prey animals that were brought to ash sites by their predators, such as a Bobcat (*Lynx rufus*) carrying a living California Vole (*Microtus californicus*) onto an ash pile. If multiple species appeared in a single image, we recorded the presence of each species as a distinct visit. In the 146 visits in which species differentiation was not possible, we classified animals as precisely as possible to genus, family, or order.

We calculated odds ratios (ORs) and 95% confidence intervals and used a two-tailed Fisher's Exact Test to determine whether animals showed a bathing preference for each ash type. To account for multiple comparisons, we adjusted P values using the Benjamini-Hochberg correction, considering $P \leq 0.05$ as the threshold for significance. In addition to ash type, we examined whether ash bathing preferences varied by site using a two-tailed Fisher's Exact Test. We also assessed whether ash-bathing behavior changed over time since ash deployment by fitting Logistic Regression models with time since deployment as a predictor and including an interaction with species to account for potential species-specific differences in temporal response. We used R version 4.3.2 (R Core Team, 2024) for all statistical analyses.

RESULTS

Wildlife visits.—There were 1,674 wildlife visits across 1,437 camera days (Appendix Table 1). Among the 27 species we identified, the most common visitors were Dark-eyed Juncos (*Junco hyemalis*), which we found at all four sites (386 visits, 23.1% of all animal visits; Appendix Table 2), Western Gray Squirrels (334 visits, 20.0% of all animal visits), and Wild Turkeys (325 visits, 19.4% of all animal visits). These three species together accounted for 62.4% of all site visits by animals.

Ash bathing.—We observed 45 independent ash bathing events. Western Gray Squirrels bathed in ash 31 times (9.6% of the 334 total Western Gray Squirrel visits), Wild Turkey 10 times (3.1% of the 325 Wild Turkey visits), and there was one event each by a Coyote (*Canis latrans*; 1.6% of the 62 Coyote visits), Striped Skunk (*Mephitis mephitis;* 1.6% of 63 Skunk visits), Dark-eyed Junco (0.3% of 386 Dark-eyed Junco visits), and Mourning Dove (*Zenaida macroura*; 11.1% of 9 Mourning Dove visits).

Of 334 total visits to ash by Western Gray Squirrels, 124 (37.1%) were to California Bay Laurel, 110 were to Coast Redwood (32.9%), and 100 were to Tasmanian Blue Gum Eucalyptus (29.9%). Squirrels were observed bathing in Coast Redwood ash (12, 10.9% of visits to Coast Redwood), Tasmanian Blue Gum Eucalyptus (10, 10.0% of visits to Tasmanian Blue Gum Eucalyptus), and California Bay Laurel (9, 7.3% of visits to California Bay Laurel; Fig. 3). These differences were not significant, however (Table 1), and there was no clear preference for a specific ash type.

TABLE 1. Pairwise comparisons of ash-bathing preferences among Western Gray Squirrel (*Sciurus griseus*) and Wild Turkey (*Meleagris gallopavo*) visits, with adjusted (Adj.) P values using the Hochberg correction for multiple comparisons. None of the comparisons were significant. Plant comparisons for each animal species are CBL = California Bay Laurel (*Umbellularia californica*), TBGE = Tasmanian Blue Gum Eucalyptus (*Eucalyptus globulus*), CR = Coast Redwood (*Sequoia sempervirens*). The abbreviation CI = confidence interval.

Comparison	Odds Ratio (95% CI)	Adj. *P* value
Western Gray Squirrels		
CBL / TBGE	0.71 (0.24–2.02)	0.960
CBL / CR	0.64 (0.23-1.74)	0.960
TBGE / CR	0.91 (0.33–2.42)	1.000
Wild Turkeys		
CBL / TBGE	0.00 (0.00–1.08)	0.110
CBL / CR	0.00 (0.00–1.69)	0.248
TBGE / CR	0.76 (0.15–3.30)	0.755

Of 325 visits to ash by Wild Turkeys, 123 were to Coast Redwood (37.8%), 107 to Tasmanian Blue Gum Eucalyptus (32.9%), 95 to California Bay Laurel (29.2%). They were observed ash bathing in Coast Redwood (6, 4.9% of visits to Coast Redwood) and in Tasmanian Blue Gum Eucalyptus (4, 3.7% of visits to Tasmanian Blue Gum Eucalyptus), but not in California Bay Laurel (Fig. 3). Overall, type of ash was not a significant factor (Table 1).

While all 12 cameras observed at least one instance of ash bathing, there were differences in how *many animals* engaged in ash bathing across sites. There were 517 total visits at Site A, 315 at Site B, 373 at Site C, and 469 at

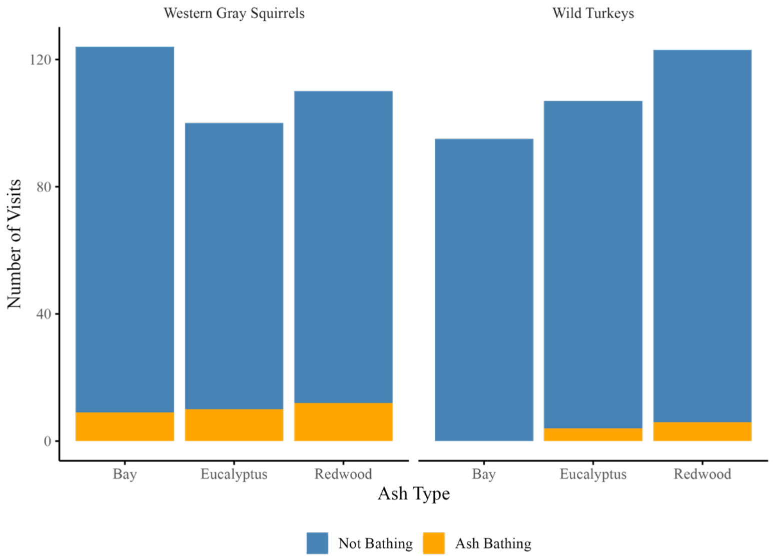

FIGURE 3. Independent visits by Western Gray Squirrels (*Sciurus griseus*) and Wild Turkeys (*Meleagris gallopavo*) at ash piles of three tree species at Quail Ridge Reserve, California from June-October 2024. Abbreviations are Bay = California Bay Laurel (*Umbellularia californica*), Eucalyptus = Tasmanian Blue Gum Eucalyptus (*Eucalyptus globulus*), and Redwood = Coast Redwood (*Sequoia sempervirens*). No significant difference in visit frequency to ash piles from the different tree species was detected ($P > 0.05$).

Table 2. Site characteristics where camera trapping surveys of wildlife activity were conducted June-October 2024. Tree count reflects number of trees > 7.6 cm (3.0 inches) diameter at breast height within 2 m of the camera.

Camera	Tree Count	Dominant Oak Species	Canopy Coverage	Notes
Site A				
1	4	Live Oak	38.38%	Near marina; low tree density and canopy cover
2	1			
3	1			
Site B				
1	2	Black Oak	60.56%	Site bisected by road; closest to water; relatively dense leaf litter
2	1			
3	1			
Site C				
1	4	Live Oak	63.17%	Dense understory with hip-level vegetation; wild grapes and berry-producing plants present; abundant coarse woody debris; located in floodplain with moist-soil species; high acorn density and squirrel caching activity
2	2			
3	2			
Site D				
1	3	Live Oak	55.32%	Sparse shrub layer; evidence of historical road disturbance; grape vines present
2	5			
3	1			

Site D. Of all ash-bathing events, 42.2% occurred at Site C, 26.7% at Site A, 24.4% at Site D, and 6.7% at Site B. Metrics of habitat quality varied between sites, with Site C featuring the highest tree canopy coverage and highest density of hip-level vegetation (Table 2). Western Gray Squirrels bathed during 13.5% of visits at Site C (13/96), 7.1% at Site D (7/98), 9.2% at Site A (11/119), and never at Site B (0/21). These differences were not significant (P = 0.629 in all comparisons). Wild Turkeys bathed during 7.7% of visits at Site C (6/78), 4.4% at Site D (3/69), 1.0% at Site A (1/97), and never at Site B (0/81). There were no significant preferences of one site over another (Table 3). Visual inspection of the 45 ash-bathing events did not suggest any temporal variation in bathing behavior (Fig. 4). Time since deployment was not a significant predictor of bathing probability for Wild Turkeys and Western Gray Squirrels ($F_{1,657} = 0.174$, $P = 0.679$). Including an interaction between time and species to assess species-specific trends also showed no significant effect ($F_{1,655} < 0.001$, $P = 0.993$), indicating that temporal trends were similar across both species.

Table 3. Adjusted pairwise Fisher's Exact Test P values comparing Wild Turkey (*Meleagris gallopavo*) ash bathing preferences between sites.

	Site B	Site C	Site D
Site C	0.076	–	–
Site D	0.380	1.000	–
Site A	1.000	0.229	0.926

Discussion

Our study provides the first systematic documentation of ash bathing behavior in free-ranging Western Gray Squirrels and Wild Turkeys and reveals previously undescribed patterns in both the frequency and context of this behavior. Among 23 vertebrate species detected across four study sites, only six were observed engaging in ash bathing, with Western Gray Squirrels and Wild Turkeys accounting for the majority of these events. In fact, Western Gray Squirrels bathed in ash during nearly 10% of their site visits. We did not observe any preferences among Western Gray Squirrels or Wild Turkeys for the three ash types offered. Prior

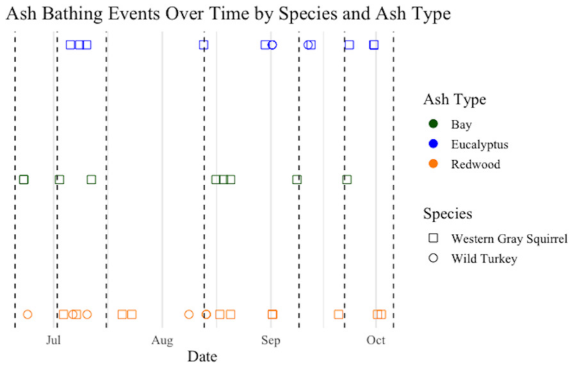

Figure 4. Timing of ash bathing events among dominant ash-bathing species following ash deployment (indicated by dashed vertical lines). Points represent individual bathing events, colored by ash type and shaped by species.

studies have shown that substrate preferences for dust bathing in Wild Turkeys and other Galliformes are variable, though fine, dry, loose particles like sand are often preferred (Olsson and Keeling 2005; de Jong et al. 2007; Wang 2018; Monckton 2020). Most accounts for squirrel species describe the use of dust or sand (Steiner 1974; Long and Smith 2023); however, chipmunks have been reported bathing in fine bark debris from Giant Sequoia (*Sequoiadendron giganteum*; Johnston 1998). All three ash types offered in our study had similar consistency, having been crushed into a powder and dried before deployment.

Both Western Gray Squirrels and Wild Turkeys serve as hosts for the Western Blacklegged Tick, the primary vector of *Borrelia burgdorferi*, the bacterium the causes Lyme disease (Lane et al. 2006; Salkeld et al. 2008), and *Anaplasma phagocytophilum*, the agent of granulocytic anaplasmosis (Rejmanek et al. 2011). Western Gray Squirrels, in particular, are key reservoir hosts for both pathogens (Lane et al. 2005; Salkeld et al. 2008; Nieto and Foley 2008; Foley et al. 2008; Rejmanek et al. 2011). Their demonstrated willingness to bathe in Tasmanian Blue Gum Eucalyptus ash, and in the case of squirrels, also California Bay Laurel ash, shows that these hosts engage in ash bathing with ash types that have known acaricidal effects against the Western Blacklegged Tick (Rubino 2024), though we did not observe a preference for one type of ash over another. In laboratory trials, the majority of adult ticks died within 48 h when exposed to these ash types. The question of whether this behavior effectively reduces tick burdens in wild populations should be addressed in the future.

We did not observe ash bathing in other wildlife hosts present in the study area that serve as primary hosts for tick species, such as Mule Deer (*Odocoileus hemionus*; Castro and Wright 2007). An anecdote about deer (*Odocoileus* sp.) bathing in ash is available (Street Roots. 2020. Native land management could save us from wildfires, experts say. Available from https://www.streetroots.org [Accessed 20 January 2025]), however, and Mule Deer frequently visited our camera sites, where they were observed sniffing the ash. We suspect that the ash piles may have been too small for deer to use for bathing.

Habitat characteristics appeared to influence ash bathing behavior. The highest frequency of ash bathing was observed at Site C, which was situated on a floodplain, with abundant hip-level vegetation and visible squirrel activity zones including caching areas and abundant acorns (pers. obs.). Squirrels sometimes use dust bathing in chemical communication (Steiner 1974; Long and Smith 2023). Although Site C did not have the highest overall squirrel visitation, its high caching and foraging activity may have influenced how squirrels used the site, including the possibility that ash bathing served a communicative function related to territoriality or resource marking. This site also had the highest average canopy cover and densest vegetation of all the sites. While limited research is available on Wild Turkeys, other Galliformes have similarly shown a preference for dust bathing in areas with dense tree cover that offer more concealment from predators (Hein 1970; Wang et al. 2018).

Ash is produced naturally in wildfires and deliberately through controlled burns. These burns may provide an opportunity to integrate tick control by creating ash that is both acaricidal and attractive to wildlife, which can be placed at sites where key hosts are likely to bathe. This approach echoes ethnoveterinary practices in which ash bathing is used to manage ectoparasites in domestic animals (Banjo et al. 2009; Moreki 2013; Wanzala 2017), suggesting that traditional knowledge could inform strategies to enhance acaricidal efficacy in wild populations.

We recognize that our study had several limitations. Although conducted in an oak forest, we did not evaluate wildlife use of oak ash due to safety concerns regarding its carcinogenic potential (Maciejewska et al. 1993), though this remains a relevant future research avenue. Additional work in other forest types, such as Coast Redwood and Douglas Fir (*Pseudotsuga menziesii*), would help determine whether the behaviors observed here can be generalized beyond our study system. Predator scent marking, particularly by coyotes, may have influenced visitation rates and ash bathing behavior by prey species in our study (Mahr and Hoi 2018). Finally, we rotated the ash piles at each site so that every site had every type of ash. This rotation helps account for differences not due to ash type but to location and ensures that observed ash-bathing patterns reflect ash type rather than site-specific effects. Residual ash could, however, have caused cross-contamination. Nevertheless, our findings, paired with existing evidence of ash toxicity to ticks, reveal a promising intersection between wildlife behavior, vector ecology, and fire management. Although no significant preference for ash types was detected, the willingness of key tick hosts to engage in ash bathing nonetheless highlights an important avenue for future research on wildlife-mediated ectoparasite control. Future research should explore whether the strategic use of specific ash-producing vegetation in controlled burns can help suppress tick populations and reduce pathogen transmission in natural systems.

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Ryan Meadows is a Ph.D. student in the Entomology and Nematology program at the University of California, Davis. His research focus is on the use of insects in organic waste management and bioconversion. He earned a B.S. in Entomology from the same institution, where he contributed to research in several labs, studying arthropods in agriculture, ecology, and vector control. (Photographed by Ryan Meadows).

Francesca Rubino is a Disease Ecologist who uses statistical modeling as well as quantitative field and laboratory studies to investigate the ecology and epidemiology of tick-borne diseases. She earned her B.S. in Conservation and Resource Studies and a M.S.P.H. in Global Health and the Environment from the University of California, Berkeley, and her Ph.D. in Epidemiology from the University of California, Davis. Francesca aims to apply her work to integrate public health and ecological conservation efforts to promote healthier ecosystems and communities. (Photographed by Anthony Parker).

Kailyn Lozano is a Ph.D. graduate student in the interdisciplinary Comparative Health Sciences program, administered by the Carlson College of Veterinary Medicine at Oregon State University, Corvallis. Her current research focuses on disease ecology, particularly Crimean-Congo hemorrhagic fever and Foot-and-Mouth disease. Kailyn earned a B.A. in Evolution, Ecology, and Biodiversity, with a minor in Global Disease Biology from the University of California, Davis. While there, she contributed to research in an Infectious Disease Ecology Lab, studying the ecology of tick-borne diseases. (Photographed by Drew Morganstein).

Austin Roy is a Small Mammal Ecologist who studies how wildlife interact with their environment and shape ecosystems and seeks to understand how species management affects these interactions. He earned his B.S. in Wildlife from Humboldt State University in Arcata, California, and his Ph.D. in Ecology and Evolutionary Biology from the University of Texas at El Paso. Austin aims to apply his expertise to the conservation of wildlife and natural resources for the benefit of the public. (Photographed by Austin Roy).

Janet Foley is a Professor and Researcher for the Department of Veterinary Medicine and Epidemiology in the School of Veterinary Medicine at the University of California, Davis. She studies the ecology and epidemiology of infectious diseases in complex communities. Research in her lab aims to understand how community complexity contributes to disease persistence and emergence, and how driving factors are affected by anthropogenic change. (Unknown photographer).

APPENDIX

APPENDIX TABLE 1. Behaviorally categorized independent visits by wildlife species to in front of camera stations at Quail Ridge Reserve in Napa County, California, from June-October 2024. Abbreviations are Bay = California Bay Laurel (*Umbellularia californica*), Eucalyptus = Tasmanian Blue Gum Eucalyptus (*Eucalyptus globulus*), and Redwood = Coast Redwood (*Sequoia sempervirens*).

Behavior Category	Species	Scientific name	Bay	Eucalyptus	Redwood	Total
Bathing in ash	Coyote	*Canis latrans*	1			1
	Dark-eyed Junco	*Junco hyemalis*	1			1
	Mourning Dove	*Zenaida macroura*			1	1
	Striped Skunk	*Mephitis mephitis*		1		1
	Western Gray Squirrel	*Sciurus griseus*	9	10	12	31
	Wild Turkey	*Meleagris gallopavo*		4	6	10
	Bathing Total		11	15	19	45
Foraging on ash	Bobcat	*Lynx rufus*	1		2	3
	Coyote	*Canis latrans*	15	5	5	25
	Dark-eyed Junco	*Junco hyemalis*	9	6	4	19
	Mule Deer	*Odocoileus hemionus*	16	15	15	46
	Gray Fox	*Urocyon cinereoargenteus*	4	3	3	10
	Mourning Dove	*Zenaida macroura*			1	1
	Mouse spp.	*Mus* spp.			1	1
	California Quail	*Callipepla californica*			1	1
	Striped Skunk	*Mephitis mephitis*	5		2	7
	Western Gray Squirrel	*Sciurus griseus*	23	15	16	54
	Wild Turkey	*Meleagris gallopavo*	9	13	10	32
	Unidentified Passerine	—		3	2	5
	Foraging on Ash Total		82	60	62	204
No Ash Interaction	Bobcat	*Lynx rufus*	2	4	4	10
	Canada Goose	*Branta canadensis*	1	1		2
	Coyote	*Canis latrans*	13	5	6	24
	Dark-eyed Junco	*Junco hyemalis*	94	95	99	288
	Mule Deer	*Odocoileus hemionus*	41	43	30	114
	Domestic Dog	*Canis lupus familiaris*		1		1
	Gray Fox	*Urocyon cinereoargenteus*	15	18	9	42
	Frog spp.	—		1	2	3
	Botta Pocket Gopher	*Thomomys bottae*		1		1
	House Finch	*Haemorhous mexicanus*		2	1	3
	Magpie spp.	*Pica* spp.			1	1
	Mountain Lion	*Puma concolor*	1		1	2
	Mourning Dove	*Zenaida macroura*	1	3	2	6
	Mouse spp.	*Mus* spp.	3	4	7	14
	Virginia Opossum	*Didelphis virginiana*		2	4	6
	California Quail	*Callipepla californica*	1	1	2	4
	Red Shouldered Hawk	*Buteo lineatus*		1		1
	Striped Skunk	*Mephitis mephitis*	12	13	16	41
	Spotted Towhee	*Pipilo maculatus*	2	2	3	7
	Western Gray Squirrel	*Sciurus griseus*	70	66	65	201

Appendix Table 1 (continued). Behaviorally categorized independent visits by wildlife species to in front of camera stations at Quail Ridge Reserve in Napa County, California, from June-October 2024. Abbreviations are Bay = California Bay Laurel (*Umbellularia californica*), Eucalyptus = Tasmanian Blue Gum Eucalyptus (*Eucalyptus globulus*), and Redwood = Coast Redwood (*Sequoia sempervirens*).

Behavior Category	Species	Scientific name	Bay	Eucalyptus	Redwood	Total
	Hermit Thrush	*Catharus guttatus*	5	11	7	23
	Wild Turkey	*Meleagris gallopavo*	60	68	72	200
	Unidentified Passerine	—	17	29	21	67
	California Vole	*Microtus californicus*	2	2	2	6
	Western Bluebird	*Sialia mexicana*	2	1		3
	Western Fence Lizard	*Sceloporus occidentalis*	1	1	3	5
	Western Screech Owl	*Megascops kennicottii*		3	2	5
	Western Wood Pewee	*Contopus sordidulus*			1	1
	White Breasted Nuthatch	*Sitta carolinensis*			1	1
	No Ash Interaction Total			378	361	1082
Sitting on ash	Bobcat	*Lynx rufus*	2		2	4
	Dark-eyed Junco	*Junco hyemalis*	1		1	2
	Gray Fox	*Urocyon cinereoargenteus*		1		1
	Western Gray Squirrel	*Sciurus griseus*			2	2
	Wild Turkey	*Meleagris gallopavo*	1	1		2
	Sitting on Ash Total		4	2	5	11
Standing on ash	Cooper's Hawk	*Accipiter cooperii*		1	1	2
	Dark-eyed Junco	*Junco hyemalis*	4	3	9	16
	Mule Deer	*Odocoileus hemionus*	3		2	5
	Domestic Dog	*Canis lupus familiaris*			1	1
	Gray Fox	*Urocyon cinereoargenteus*	2	1	2	5
	Spotted Towhee	*Pipilo maculatus*			1	1
	Western Gray Squirrel	*Sciurus griseus*	2	1	2	5
	Hermit Thrush	*Catharus guttatus*	2			2
	Wild Turkey	*Meleagris gallopavo*	3			3
	Unidentified Passerine	—	2	2		4
	Standing on Ash Total		18	8	18	44
Walking on ash	Bobcat	*Lynx rufus*	1	1		2
	Canada Goose	*Branta canadensis*		1		1
	Coyote	*Canis latrans*	10		2	12
	Dark-eyed Junco	*Junco hyemalis*	18	12	30	60
	Mule Deer	*Odocoileus hemionus*	7	3	16	26
	Domestic Dog	*Canis lupus familiaris*	1			1
	Gray Fox	*Urocyon cinereoargenteus*	6	9	6	21
	Frog spp.	—		1		1
	House Finch	*Haemorhous mexicanus*			1	1
	Mountain Lion	*Puma concolor*	1			1
	Mourning Dove	*Zenaida macroura*			1	1
	Mouse spp.	*Mus* spp.	1	3	1	5
	Virginia Opossum	*Didelphis virginiana*		1	1	2
	California Quail	*Callipepla californica*			3	3
	Striped Skunk	*Mephitis mephitis*	5	6	3	14
	Spotted Towhee	*Pipilo maculatus*	2		1	3

Appendix Table 1 (continued). Behaviorally categorized independent visits by wildlife species to in front of camera stations at Quail Ridge Reserve in Napa County, California, from June-October 2024. Abbreviations are Bay = California Bay Laurel (*Umbellularia californica*), Eucalyptus = Tasmanian Blue Gum Eucalyptus (*Eucalyptus globulus*), and Redwood = Coast Redwood (*Sequoia sempervirens*).

Behavior Category	Species	Scientific name	Bay	Eucalyptus	Redwood	Total
	Western Gray Squirrel	*Sciurus griseus*	20	8	13	41
	Hermit Thrush	*Catharus guttatus*	1			1
	Wild Turkey	*Meleagris gallopavo*	22	21	35	78
	Unidentified Passerine	—		5	6	11
	Western Bluebird	*Sialia mexicana*	1			1
	Dusky-footed Woodrat	*Neotoma fuscipes*		1	1	2
	Walking on Ash Total		96	72	120	288
	Total		554	535	585	1,674

Appendix Table 2. Totaled independent visits by wildlife species to tree ash deposited in front of camera stations at Quail Ridge Reserve in Napa County, California from June-October 2024. Bay = California Bay Laurel, Eucalyptus = Tasmanian Blue Gum Eucalyptus, and Redwood = Coast Redwood. Scientific names are given in Appendix Table 1.

Species	Bay	Eucalyptus	Redwood	Grand Total	% of Total Visits
Dark-eyed Junco	127	116	143	386	23.06
Western Gray Squirrel	124	100	110	334	19.95
Wild Turkey	95	107	123	325	19.41
Mule Deer	67	61	63	191	11.41
Gray Fox	27	32	20	79	4.72
Striped Skunk	22	20	21	63	3.76
Coyote	39	10	13	62	3.7
Hermit Thrush	8	11	7	26	1.55
Mouse spp.	4	7	9	20	1.19
Bobcat	6	5	8	19	1.14
Spotted Towhee	4	2	5	11	0.66
Mourning Dove	1	3	5	9	0.54
Virginia Opossum		3	5	8	0.48
California Quail	1	1	6	8	0.48
California Vole	2	2	2	6	0.36
Western Fence Lizard	1	1	3	5	0.3
Western Screech Owl		3	2	5	0.3
Western Bluebird	3	1		4	0.24
Frog spp.		2	2	4	0.24
House Finch		2	2	4	0.24
Mountain Lion	2		1	3	0.18
Canada Goose	1	2		3	0.18
Domestic Dog	1	1	1	3	0.18
Cooper's Hawk		1	1	2	0.12
Dusky-footed Woodrat		1	1	2	0.12
Botta Pocket Gopher		1		1	0.06
Magpie spp.			1	1	0.06
Red Shouldered Hawk		1		1	0.06
Western Wood Pewee			1	1	0.06
White Breasted Nuthatch			1	1	0.06
Unidentified Passerine	19	39	29	87	5.2
Grand Total	554	535	585	1,674	

2025 Annual Meeting Review

72nd Annual Meeting

A RETURN TO NATURAL HISTORY

The Western Section of The Wildlife Society
72nd Annual Meeting

Visalia Convention Center
February 3-7, 2025

Program Chair: Jeff Alvarez, The Wildlife Project

Attendance: 792 participants including 192 students and 61 Early Career Professionals.

Plenary Theme: A Return to Natural History

Plenary Speakers:

- **Robert Hansen**, Herpetologist

Keynote Address - *Extreme Conservation*

- **Dr. Joel Berger,** Colorado State University

Capstone Presentation:

- **Dr. Earyn McGee,** Herpetologist – Science Communicator – #Afro_Herper

2025 Pre-Conference Workshops:

- Small Mammal Symposium
- Training for High-Speed Rail Jobs
- Wildlife Biologist Construction Awareness Training (WildC.A.T.)
- CyberTracker One-day Certification
- Introduction to Wildlife Tracking
- Visalia Field Trips

Student Presentation Awards:

Oral Presentation Awards

1st Nicole Lindenauer, UC Davis: Augmenting Translocated Sage-grouse Broods & Guiding Future Release Locations

2nd: Tali Caspi, UC Davis: Impervious surface cover and number of restaurants shape diet variation in an urban carnivore

3rd: Margaret Mercer, University of Arizona: Why did the bobcat cross the road? Urban bobcat behavior and roadkill mitigation strategies

Poster Presentation Awards

1st: Destiny Saucedo, Cal Poly Humboldt: Spatial Dynamics of San Clemente Island Fox Adult Females and Pups

2nd: Deirdre Replinger, Cal Poly Humboldt: Pacific martens, a forest obligate, persisted after a megafire

3rd: Sierra Olsen, Cal Poly Humboldt: Nest Site Selection of Ferruginous hawk within Butte Valley

In Memoriam

- **Patrick Lieske 1971-2024**
- **Ron Duke 1951-2024**

2025 Awards Bestowed:

- The Raymond F. Dasmann Award for the Professional of the Year went to **Dan Airola**
- The Conservationist of the Year Award went to the **Fresno Chaffee Zoo**
- The Chapter of the Year Award went to the **Southern California Chapter**
- The James D. Yoakum for Outstanding Service and Commitment to The Western Section of The Wildlife Society went to **Candace Renger**

2025 Western Section Workshops:

- **2-Part Diversity, Anti-Racism & Inclusion Training,** April 14 & 16, 2025 AND September 30 & October 2, 2025, Online
- **Western Spadefoot Workshop,** April 17, 20-21, 2025 in Livermore, CA
- **Flat-Tailed Horned Lizard Workshop**, May 13, 2025 in Yuma, AZ
- **Advanced Bat Acoustics with Joe Szewczak,** June 13-15, 2025, Online
- **Marine Mammal Identification,** July 17-20, 2025 in Santa Barbara, CA
- **Bumble Bee Field Course,** July 29-August 1, 2025 in Truckee, CA
- **W+oW Resource Retreat,** October 3-5, 2025 in Bodega Bay, CA
- **Western Monarch Overwintering Ecology & Habitat,** November 12-13, 2025 in Santa Barbara, CA

2025 Western Section Membership

Retired: 48
Regular: 749
Lifetime: 102
Contributing: 5

Student: 254
New Professional: 146
Supporting: 6
Honorary: 39*

Total: 1310
(does not include Honorary)

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President **Jeff Alvarez** The Wildlife Project	*Past-President* **Brooke Langle** SWCA	*President-Elect* **Brock Ortega** Dudek
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