WESTERN WILDLIFE

THE JOURNAL OF THE WESTERN SECTION OF THE WILDLIFE SOCIETY

California Red-legged Frog (*Rana draytonii*), Wragg Ridge Preserve, Napa County, California. (Photographed by Jeff A. Alvarez).

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2024 Peer Reviewers

The Editors of *Western Wildlife* would like to acknowledge the following biologists for their expert reviews during the 2024 publication year. Your efforts, dedication, and professionalism have ensured that *Western Wildlife* continues to be an outlet for the ecology, natural history, management, and conservation biology of animals in western North America.

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Notes

Triploid Parthenogenetic *Aspidoscelis neotesselatus* (Colorado Checkered Whiptail): Persistence in Fragmented Urban Habitat

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Abstract.—Triploid parthenogenetic *Aspidoscelis neotesselatus* (Colorado Checkered Whiptail), the only endemic reptile in Colorado, USA, has a natural geographic distribution that is limited to a small area in the southeastern part of the state. Nevertheless, the species is highly variable with four distinctive color pattern classes designated A, B, C, and D. We undertook this study to determine the ecological status of hybrid-derived *A. neotesselatus* at the Chain of Lakes (COL) area in the city of Pueblo, Pueblo County. This narrow component of Lake Pueblo State Park, situated between a busy urban highway and the Arkansas River, comprises the smallest and most fragmented acreage of habitation known to us for this squamate. A visit to COL by LJL ca. 20 years after discovery and collection of the species there by JMW in September 1999 and June 2000 revealed that it remains abundant at the site based on 17 lizards observed between 0813 h and 0950 h on 18 June 2021.

Key Words.—abundance; Colorado; fragmented habitat; parthenogenetic reproduction; whiptail lizards

Resumen.—*Aspidoscelis neotesselatus* (Huico Teselado de Colorado), triploide partenogenético, es el único reptil endémico de Colorado, EE.UU., tiene una distribución geográfica natural que se limita a una pequeña área en la parte sureste del estado. Sin embargo, la especie es muy variable según cuatro clases de patrones de color distintivos designados A, B, C y D. Llevamos a cabo este estudio para determinar el estado ecológico de *A. neotesselatus* derivado de híbridos en el área de la Cadena de Lagos (COL) en la ciudad de Pueblo, Condado de Pueblo. Este estrecho componente del Parque Estatal Lake Pueblo, situado entre una transitada carretera urbana y el Río Arkansas, comprende la superficie habitada más pequeña y fragmentada que conocemos para esta lagartija. Una visita a COL por LJL ca. 20 años después del descubrimiento y recolección de la especie en ste sitio por parte de JMW en septiembre de 1999 y junio de 2000, reveló que sigue siendo abundante en el sitio segúnd.

Palabras Clave.—abundancia; Colorado; hábitat fragmentado; largartos cola de látigo; reproducción parthenogenética

The taxonomic and nomenclatural status of Colorado Checkered Whiptail (*Aspidoscelis neotesselatus*) is based on the following studies: report of its existence though included in *Cnemidophorus* = *Aspidoscelis tesselatus* as pattern classes A and B (Zweifel 1965), description of the triploid species (Walker et al. 1997), generic status (Reeder et al. 2002), and grammatical implications of the generic name *Aspidoscelis* on the suffixes of species names (Tucker et al. 2016). We have studied this triploid parthenogenetic lizard in most of the areas within its small natural geographic distribution in southeastern Colorado, USA, in parts of Crowley, El Paso, Fremont, Huerfano, Las Animas, Otero, Pueblo, and Teller counties. We note that the adaptability of the species, which is likely one major advantage of parthenogenetic reproduction (Taylor and Livo 2023), is indicated by presence of the following introduced arrays (= groups): Colorado in Denver and Adams counties (Livo et al. 2019, 2022), Douglas County (Taylor et al. 2015b; Livo et al. in press), and in distant Grant County, Washington (Weaver et al. 2011). The biology of the species has been most intensively studied in either Pueblo County (e.g.,

Knopf 1966; Taylor et al. 2006; Walker 2012) or it and Otero counties (e.g., Parker and Selander 1976; Walker et al. 1995, 1997, 2012; Taylor et al. 2015a). Interstate Highway 25 seems to be the dividing line between areas in the city of Pueblo conducive to continuing success of *A. neotesselatus* to the west and areas where it apparently has been extirpated to the east (Walker et al. 1996; Walker, unpubl. data). Sites west of Interstate 25 either very near Lake Pueblo State Park (e.g., Nature and Raptor Center of Pueblo) or within the park proper (e.g., Juniper Breaks Campground, Arkansas Point Campground, Park Headquarters, Chain of Lakes, and Remote Launch Ramp) support this species in abundance (Walker et al. 1997; Walker 2012) and are critical to its conservation status. Although little human activity takes place at the abandoned Remote Launch Ramp site bordering Pueblo Lake, elsewhere in the area *A. neotesselatus*, which is not noted for its wariness, carries on normal diurnal activities in campground and scenic areas intensively used by humans who are either oblivious to or tolerant of the presence of this all-female squamate (pers. obs.).
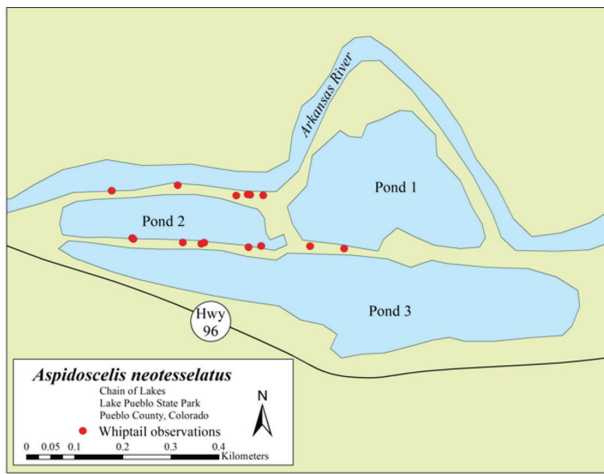
FIGURE 1. Map of Chain of Lakes area of Lake Pueblo State Park, Pueblo County, Colorado, showing points of observations and images of 17 triploid parthenogenetic *Aspidoscelis neotesselatus* A made 18 June 2021 between 0813 and 0950 (multiple lizards were observed at some waypoints).

Of the sites of occurrence mentioned for *A. neotesselatus* in Pueblo County (Walker et al. 1997 and herein), the habitat used by *A. neotesselatus* in the Chain of Lakes (COL) area constitutes the most unusual known for the species. One of us (JMW) serendipitously discovered the species there 6 September 1999 when only young-of-year (YOY) were active in what was then known as Valco Ponds State Wildlife Area. A visit to the site the next day also revealed only YOY; however, on 10 June 2000, JMW observed several year classes including gravid females. The purpose of the recent visit to COL was to assess the status of *A. neotesselatus* there over 20 y after its discovery and collection (see Walker 2012).

The COL component is narrowly sandwiched between the north side of Colorado Highway 96 (i.e., Thatcher Avenue) and the Arkansas River west of metropolitan

Pueblo, Pueblo County. The site (38.259361°N, 104.705824°W, WGS84; elevation 1,445 m) is on the opposite side of the river from the Nature and Raptor Center of Pueblo, which is also inhabited by *A. neotesselatus*. The COL site was purchased by the state in 2005 and developed as a public wildlife sanctuary and recreational area (Fig. 1). It was subsequently incorporated into Lake Pueblo State Park. We have no evidence that the state was aware of the importance of the site to the Colorado endemic *A. neotesselatus* per se at the time of the property transfer. Surveys conducted in 2010 documented occurrences of *A. neotesselatus* in parts of the state park, and while the COL area was identified as habitat for this species, no records were reported at that time (Clinte Henke, pers. comm.). Although the entirety of COL encompasses about 104 ha and features seven ponds, five stocked with species of game fishes and available to the public, as little as about 2.1 ha of this total acreage appears to be suitable habitat for *A. neotesselatus* (Fig. 2), which occurs there in the absence of gonochoristic congener Prairie Racerunner (*A. sexlineatus viridis*). A management plan for COL has been proposed. The ponds, wetlands, and wildlife areas are mostly either reclaimed gravel pits or their surroundings. Ponds 1–3 are separated from the Arkansas River (Fig. 1) by a narrow band of habitat of about 20–30 m in width used by whiptail lizards (Fig. 2).

We here provide the results of a visit to COL 18 June 2021. During the visit between 0813 and 0950, we took photographic vouchers of 17 *A. neotesselatus*, including adults and pre-reproductive individuals (Fig. 3). The route included the narrow trail between ponds 2 and 3, a short distance between ponds 1 and 3, and the area north of Pond 2 between it and the Arkansas River, a total distance of approximately 1.4 km. The included images of lizards in situ at COL (Fig. 3) provide evidence of dorsal color and pattern variability based on ontogenetic
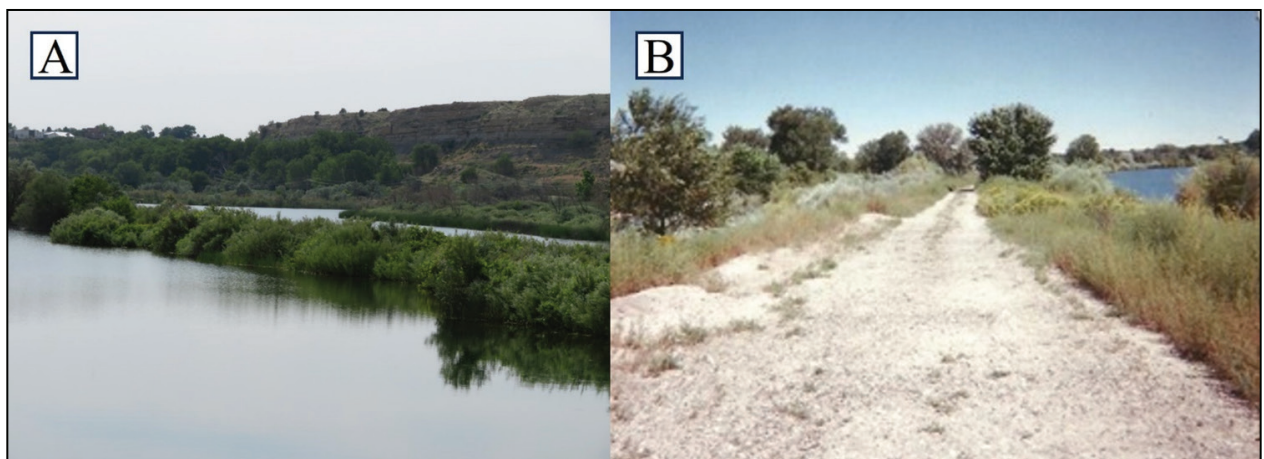
FIGURE 2. (A) View looking to the southeast 19 June 2021, showing narrow vegetated area inhabited by *Aspidoscelis neotesselatus* A between ponds 2 and 3 (Fig. 1) of Chain of Lakes (COL), Pueblo, Pueblo County, Colorado. (Photographed by Lauren Livo). (B) Looking eastward 6 September 1999 on the narrow roadway on the levee between the Arkansas River on the left and a pond on the right at COL, Pueblo, Pueblo County, Colorado, featuring highly productive habitat for *A. neotesselatus* A in September 1999 and June 2000. (Photographed by James Walker).

FIGURE 3. (A) Young adult *Aspidoscelis neotesselatus* A, relative age indicated by uninterrupted lateral stripe and pattern of spots, on relatively open substrate between ponds 2 and 3 (Fig. 1) at Chain of Lakes (COL), Pueblo, Pueblo County, Colorado. (B) Older adult *A. neotesselatus* A, relative age indicated by interrupted lateral stripe and pattern of many spots, on relatively open substrate between a pond and the Arkansas River at COL. (C) Pre-reproductive *A. neotesselatus* A, relative age status indicated by body size and dorsal color pattern of lizard, on debris-cluttered substrate at COL. (D) Pre-reproductive *A. neotesselatus* A, relative age status indicated by body size and dorsal color pattern of lizard, on debris-cluttered substrate at COL. (Photographed by Lauren Livo).

and individual variation for triploid parthenogenetic *A. neotesselatus* based on specimens previously collected. The images also depict the substrate characteristics of habitat frequented by lizards at COL in 2021.

In 1999 and 2000, we found *A. neotesselatus* A at COL in large numbers (unpubl. data). In those years, lizards were present along the approximately 3 m wide by 300 m long forest trail running west from the parking lot along the Arkansas River. The only exposed substrate was that of the trail, which seemed critical to the presence of lizards living in this narrow band of habitat where they were frequently forced to retreat from foraging and basking behaviors by humans. In 1999 and 2000, most of the lizards observed were located east of the westerly Valco Parking Lot on the approximately 80 m long road/levee between the Arkansas River and pond 2 (Fig. 1). In 2021, more than 20 y after discovery of *A. neotesselatus* at the site, we covered additional areas east of the Valco Parking Lot and found a continued abundance of this species in a strikingly constricted habitat association.

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Peer-Edited Notes

Western Kingbird (*Tyrannus verticalis*) Feeds Pacific Chorus Frog (*Pseudacris regilla*) to Fledglings

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Abstract.—Western Kingbirds (*Tyrannus verticalis*) are migratory flycatchers that breed in spring and summer in the Western U.S. and winter in Central America. They are insect specialists that supplement their diet with arthropods and occasionally fruits and berries. Here, I report predation by a Western Kingbird on Pacific Chorus Frog (*Pseudacris regilla*) to feed to three fledglings in Sacramento County, California. Pacific Chorus Frogs are common prey for many guilds of birds in California but are not documented as prey to flycatchers. To the best of my knowledge, this is a novel observation of a Western Kingbird, an insectivorous aerial hunter, repeatedly catching and feeding Pacific Chorus Frogs to fledglings.

Key Words.—diet; fledgling; foraging; grassland; predation; prey; Tyrannidae

Western Kingbirds (*Tyrannus verticalis*) are primarily insectivorous flycatchers that occasionally eat fruits and berries (Gamble and Bergin 2020), and rarely small vertebrates such as frogs (Terres 1980). They are a migratory species that breed in spring and summer in western North America and spend winters in Central America (Gamble and Bergin 2020). The widespread and abundant Pacific Chorus Frog (*Pseudacris regilla*) is common prey to various guilds of birds such as herons, hawks, and ducks (Ethier et al. 2021). They are less common as prey for passerine birds in the Central Valley of California, the focal area of this observation, with the exception of some families such as shrikes (Laniidae) and corvids (Corvidae; Alvarez 2005; Winkler et al. 2020a,b; Olson and Titus 2022).

Western Kingbirds and Pacific Chorus Frogs are both commonly observed species at the Illa M. Collin Conservation Preserve in Sacramento County, California (Center for Natural Lands Management [CNLM] 2022). This preserve (38.5398°N, 121.2875°W, WGS 84) is a vernal pool and annual grassland landscape with small creeks and riparian areas. It is bordered primarily by residential development as well as an airport and undeveloped non-native annual grasslands. On 9 June 2023 at 0842, I observed an adult Western Kingbird feed each of its three fledglings a Pacific Chorus Frog at the preserve (Fig. 1). The three fledglings were perched on a sign adjacent to a road with open grassland on one side and riparian edge habitat on the other. They were demonstrating begging behavior, with mouths agape and frequent calls. The adult Western Kingbird would leave the fledglings for a few minutes and return with a chorus frog foraged from the slow-moving creek about 50 m away. The chorus frogs captured by the Western Kingbird were identified by their size and characteristic mask over the eye, and appeared to be adult frogs without the remnant tail appendages found on post-metamorphic frogs. The fledglings were fed one frog at

a time, with each fledgling swallowing the prey whole. The adult kingbird diligently ensured each fledgling ate one chorus frog, despite competitive begging behavior between the siblings. I observed a more dominant fledgling make several attempts to eat additional chorus frogs, unsuccessfully, before its subdominant siblings. These observations were made from within a stationary vehicle, which acted as a blind, and facilitated clear and continuous observations.

Because Western Kingbirds are migratory, they may be more likely to vary their diet depending on prey availability (Parrish 2000). This species has been documented displaying opportunistic and flexible foraging strategies on tiger beetles (Cicindelidae) and other insects (Goldberg 1979; Schultz 1983). Additionally, research by Tallamy (2019) and other ornithologists has shown that many species of birds feed their young nutrient-dense foods that may differ from a typical adult diet. Western Kingbirds, however, are widely understood to be insect specialists with one of the smallest bill sizes of North American kingbird species (Kaufmann 1992). There is a paucity of data on predation of larger vertebrate prey such as frogs and, to my knowledge, there are no accounts of this species preying on small vertebrates to feed nestlings or fledglings. A single reference stated that Western Kingbirds occasionally take tree frogs as prey, but it does not include region, frog species, or any other descriptive information (Terres 1980). There have been limited accounts of other kingbird species preying on small vertebrates. For example, Ohlendorf (1974) observed a Cassin's Kingbird (*Tyrannus vociferans*) eating a small rodent in a study in Texas. Neotropical passerine birds more often eat reptiles and amphibians (Poulin et al. 2001) and there are a few records of the closely related Tropical Kingbird (*Tyrannus melancholicus*) eating small vertebrates such as House Geckos (*Hemidactylus frenatus*; Ramirez-Fernandez et al. 2019), tree frogs
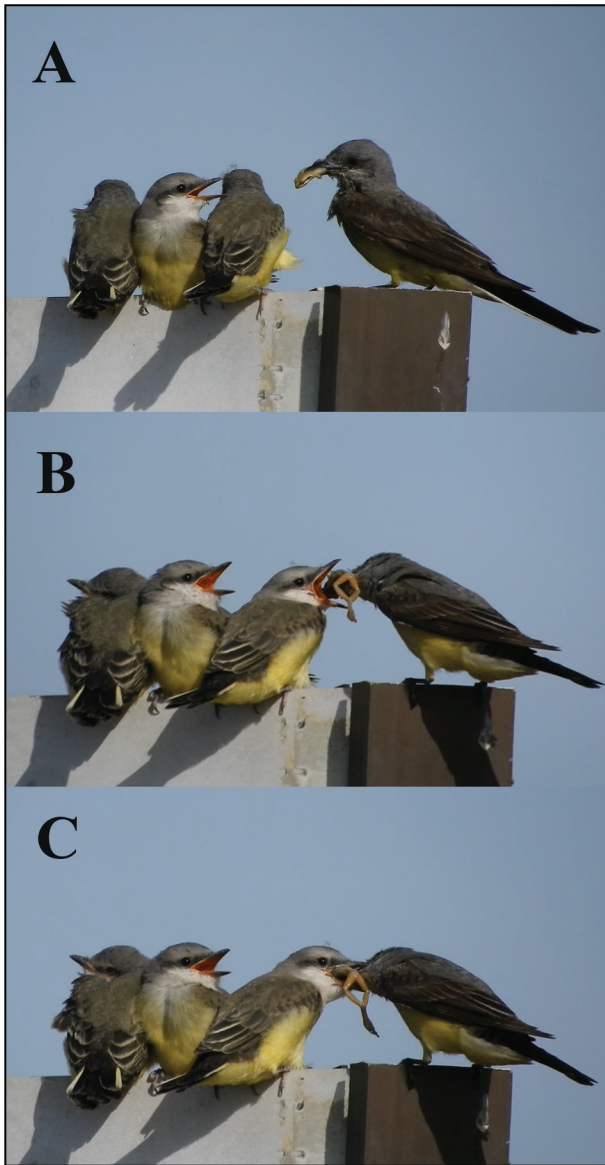
Figure 1. Western Kingbird (*Tyrannus verticalis*) feeding Pacific Chorus Frogs (*Pseudacris regilla*) to fledglings in Sacramento County, California, USA. (A) Although somewhat hard to see, there is a characteristic mask over the Pacific Chorus Frog eye. (B and C) Both siblings on the right show more aggressive begging behavior than the sibling on the left. The sibling on the left was the last to be fed by the adult. (Photographed by Allison B. Titus).

(Skutch 1954), and small fish (González-Oreja and Jiménez-Moreno 2018). Typical Western Kingbird prey such as grasshoppers, wasps, and bees (Beal 1912) are commonly observed and abundant at the preserve. Pacific Chorus Frogs are also a common and widespread species at the preserve and perhaps an easy, high nutrition prey source at this particular location (CNLM 2022). My observation of an adult Western Kingbird feeding multiple Pacific Chorus Frogs to young fledglings seems to be a novel occurrence and contributes to scant observations of Western Kingbird predation on frogs (Terres 1980). This account adds to

a more comprehensive understanding of the life-history traits of both Western Kingbirds and Pacific Chorus Frogs, and their roles in North American food webs.

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Peer-Edited Notes

Loggerhead Shrike (*Lanius ludovicianus*) Predation of a Horned Lark (*Eremophila alpestris*)

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Abstract.—The Loggerhead Shrike (*Lanius ludovicianus*) is a medium-sized songbird known for its behavior of impaling prey items. Most prey species include invertebrates such as grasshoppers and beetles, but it also takes vertebrate species such as rodents, birds, and reptiles. Herein, I report the impalement of a Horned Lark (*Eremophila alpestris*) on a barbed wire fence in San Luis Obispo County, California, likely by a Loggerhead Shrike.

Key Words.—diet; foraging; grassland; Laniidae; predation; prey

The Loggerhead Shrike (*Lanius ludovicianus*) is a medium-sized, gray songbird that commonly inhabits open landscapes with scattered shrubs, interspersed with grasses and forbs in grasslands, scrublands, steppes, deserts, prairies, and savannas throughout most of North America including Mexico (Yosef 2020). The species is infamous for impaling prey on sharp objects, such as barbed wire, cacti, thorns, and even yucca (Reid and Fulbright 1981; Yosef 2020), but it may also store its kill between the forked branches of shrubs. Once an item is impaled, the shrike typically flies off, leaving the item behind, possibly using impalement as a food cache method (Yosef and Pinshow 2005). Common prey items include invertebrates, such as grasshoppers, crickets, and beetles, and vertebrates, such as rodents, birds, amphibians, and reptiles (Cicero 1993; Clark 2011; Yosef 2020).

Of particular interest is the predation of other passerines similar in size to the Loggerhead Shrike. Bird predation is not common for the Loggerhead Shrike, which is well documented in the literature. Many of the predated birds mentioned in the literature are similar in size to the shrike and may even be larger. Birds and other vertebrate prey are typically subdued and killed by Loggerhead Shrikes by breaking the neck at the base of the skull using their hooked beak (Yosef and Pinshow 2005). Predated birds by the Loggerhead Shrike include Mourning Doves (*Zenaida macroura*; Balda 1965), Grasshopper Sparrows (*Ammodramus savannarum*; Stewart 1990), Dark-eyed Juncos (*Junco hyemalis*; Smyth 1912); Savannah Sparrows (*Passerculus sandwichensis*; Johnson 1949), and Northern Cardinals (*Cardinalis cardinalis*; Ingold and Ingold 1987). Ingold and Ingold (1987) and Tyler (1991) present a summary of avian prey species of Loggerhead Shrikes. Although Horned Lark (*Eremophila alpestris*) predation by the Loggerhead Shrike has been reported in the literature (i.e., Conley 1982; Mays 1988), only one author mentions impalement (on two spiny branchlets of the box-thorn [*Lycium*] shrub;

Wiggins 1962). Horned Larks weigh 28–48 g (Beason 2020) and likely would not be a predation challenge for the Loggerhead Shrike (weight range of 45–60 g; Yosef 2020). Herein, I describe another probable case of the impalement of a Horned Lark by a Loggerhead Shrike in San Luis Obispo County, California.

On 26 January 2011, at approximately 1230, I was driving northbound on Bitterwater Road, San Luis Obispo County, California, when I noticed a large object hanging on a barbed wire fence (Fig. 1). I pulled over onto the shoulder and walked back to find the item. Upon further examination, I identified the object as a Horned Lark. It appeared to have a broken neck and was hanging on a single barb on the fence (Fig. 1). I have observed Loggerhead Shrikes in the vicinity during previous drives along Bitterwater Road and surmised that the impaled Horned Lark was the handiwork of the shrike. I took photographs, and then left the Horned Lark in place and departed the area. The surrounding land cover was rangeland vegetated with forbs and non-native grasses.

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FIGURE 1. Horned Lark (*Eremophila alpestris*) impaled on a barbed wire fence likely by a Loggerhead Shrike (*Lanius ludovicianus*), along Bitterwater Road, San Luis Obispo County, California. (Photographed by Howard O. Clark, Jr.).

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Non-bulrush Habitat Use by Amargosa Voles (*Microtus californicus scirpensis*)

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Abstract.—Understanding how sensitive species use their habitats is critical to conservation and management efforts. The Amargosa Vole (*Microtus californicus scirpensis*) is believed to be strictly reliant on Three-square Bulrush (*Schoenoplectus americanus*, hereafter Bulrush) dominated habitats, but has anecdotally been observed in non-Bulrush dominated habitats as well. Using range-wide camera-trapping and live-trapping survey data from 2015–2016 and 2019–2020, we summarized detections of voles in non-Bulrush dominated habitats. Through live-trapping data, we observed that up to 17% of trap locations that captured voles occurred in non-Bulrush dominated habitats, with a mean distance from Bulrush habitat of 16 m. Furthermore, voles were detected at multiple camera trap locations in non-Bulrush dominated habitats. Voles were most often detected in non-Bulrush dominated habitats containing Saltgrass (*Distichlis spicata*), rushes (*Juncus* spp.), Boraxweed (*Nitrophila occidentalis*), Yerba Mansa (*Anemopsis californica*), and Common Reed (*Phragmites australis*) dominated habitats. The relatively regular detection of voles in non-Bulrush dominated habitats may indicate that these areas are also important to the ecology and biology of the species. Incorporating non-bulrush vole habitat into conservation and management objectives is likely to have multiple benefits for the conservation of the Amargosa Vole.

Key Words.—camera-trap; detection; live-trap; marsh; vegetation

Introduction

The Amargosa Vole (*Microtus californicus scirpensis*, hereafter vole) is a federally and California state-listed Endangered subspecies of the California Vole (*M. californicus*; U.S. Fish and Wildlife Service [USFWS] 2019; California Natural Diversity Database 2024). The species is only found in approximately 22 ha of disconnected marsh habitat in the Mojave Desert near Tecopa and Shoshone, California. Depending on the year, voles occur in 51–86% of available marsh sites (Deana Clifford et al., unpubl. report), with these marshes typically having low plant diversity and are dominated by Three-square Bulrush (*Schoenoplectus americanus*, hereafter bulrush), which has been positively associated with vole abundance and occupancy (Klinger et al. 2016; López-Pérez et al. 2019; Foley et al., unpubl. report). Bulrush has also been documented to comprise a dominant proportion of the diet of the vole, although bulrush cannot solely support voles, and voles must rely on a variety of different forage species (Castle et al. 2020a). As such, there has a been a misconception about the relative importance of other vegetation habitats for the vole and most management and conservation efforts have primarily focused on protecting and managing bulrush-dominated habitats. Other habitats, including bulrush-mixed habitats (López-Pérez et al. 2019), have been rarely evaluated for voles, resulting in little information on whether voles use these habitats or not. Without a comprehensive understanding of the habitat-use by the vole, we lack a complete understanding of the ecology of the species and are hindered in optimal management and conservation of it. Herein, we report on detections of voles within non-bulrush dominated habitats from various vole survey efforts.

Methods

We conducted vole surveys and vole reintroductions within the Amargosa River basin in the Mojave Desert near Shoshone (35.9797°, -116.2720°) and Tecopa (35.8824°, -116.235368°) in Inyo County, California, at elevations from 390–417 m (Fig. 1). The vole occupies wetlands fed by the Amargosa River as well as ephemeral and perennial spring-fed surface flows. The majority of marshes where voles have been studied are dominated by bulrush interspersed with other wetland plant species (e.g., graminoids, forbs) and surrounded by upland plant communities (e.g., graminoids, forbs, shrubs, and trees; Rado and Rowlands 1984).

Between 2015–2016, we live trapped small mammals using Sherman traps at 15 grid locations across the entirety of the known extant range of the vole (Janet Foley, unpubl. report). Trapping grid design followed
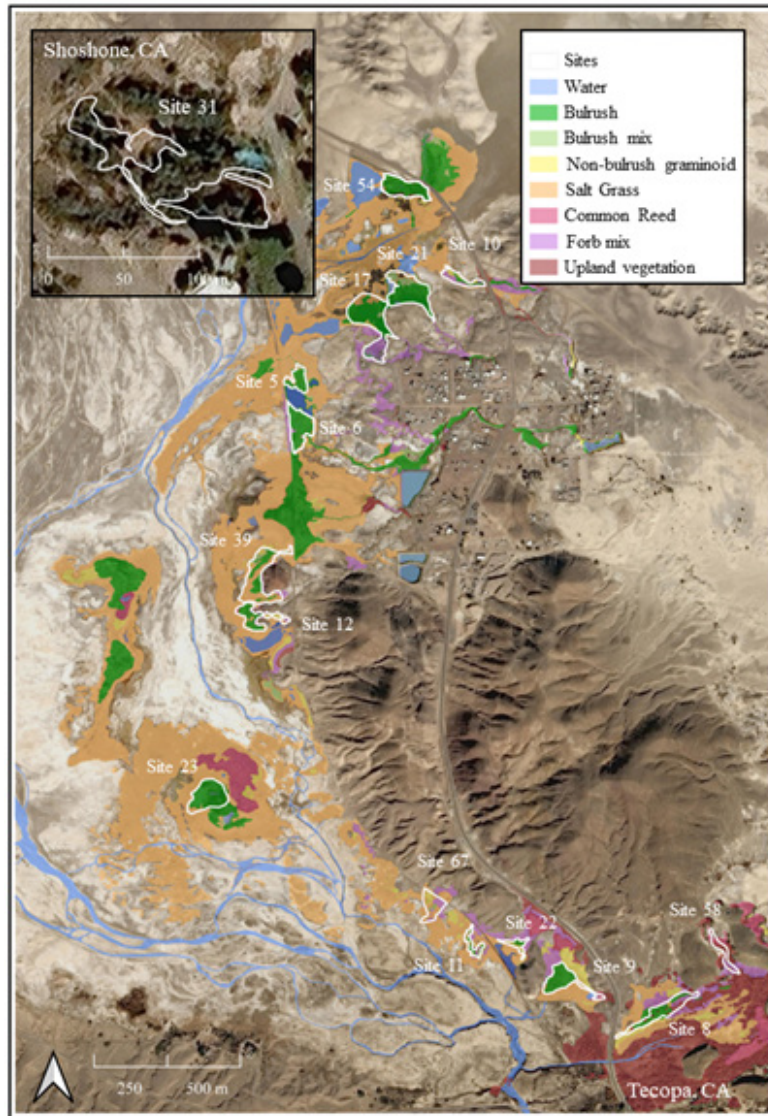
Figure 1. Map depicting major habitat types and sites surveyed for Amargosa Voles (*Microtus californicus scirpensis*) using live trapping, camera trapping, and sign surveys in 2015–2016 and 2019–2020, near Tecopa and Shoshone, Inyo County, California. Site 31 could not be included in the vegetation classification: see Site 31 description in text for details.

methodology established by Klinger et al. (2015) and each grid covered a 1-ha area, with a majority of trap locations located in bulrush-dominated habitat, but also with portions of each grid located in non-bulrush-dominated habitats. We trapped each grid for 5 d, approximately every six weeks for 12 mo. At least once during the 12-mo survey, we assessed the vegetation at each trapping location by identifying each species and quantifying the percentage cover using Daubenmire values (Daubenmire 1959; Janet Foley, unpubl. report) within a 1-m2 quadrat. To avoid sampling in areas trampled due to repeated surveys, we placed quadrats on the opposite side of the trail from each trap. Additionally, during this survey effort, we placed 1–3 baited camera traps in 21 sites, which we set to record data for approximately six weeks. We sampled most camera locations 2–3 times over the course of a year (Roy et al. 2023).

Between 2019–2020, we assessed sites for vole occupancy using un-baited camera traps at six sites and we surveyed for vole sign (feces, clipped vegetation, burrows, runways) at another seven sites. We set 14 camera traps in and on the periphery of each marsh, in areas which lacked dominant bulrush habitat. We placed camera traps in areas where sign consistent with voles was present or near burrow entrances that we suspected were occupied by voles (e.g., set at egress points from marshes to detect voles moving among marshes). The camera traps were active for 4–11 d and typically not baited, except for cameras in Site 8. We baited cameras in Site 8 with a mixture of oatmeal and peanut butter placed on the ground within the field of view of the camera. We performed sign surveys along the perimeter and areas surrounding each marsh and we recorded locations of presumptive vole sign using a GPS device. We assessed vegetation at each camera-trap location as described above.

<div style="text-align:center">RESULTS</div>

During the 2015–2016 range-wide assessment, approximately 17% of trap locations with captured voles (89/518) occurred in non-bulrush dominated habitats (< 5% cover of live bulrush or bulrush litter). At approximately half of these trap locations (53), bulrush was completely absent (0% cover) from the sampling quadrats; however, some quadrats may have been in proximity to bulrush sites (within 1 m). Across all non-bulrush dominated sites, vegetation communities consisted of > 25% cover of the following species (singly or in combination): Inland Saltgrass (hereafter saltgrass, *Distichlis spicata*, n = 35), rushes (*Juncus* spp., n = 21), Boraxweed (*Nitrophila occidentalis*, n = 6), Yerba Mansa (*Anemopsis californica*, n = 4), sedges (*Carex* spp., n = 3), Common Reed (*Phragmites australis*, n = 12), and Annual Sunflower (*Helianthus annuus*, n = 3). The distance of individual trap locations to the nearest bulrush habitat ranged from 0 m (immediately adjacent) to 61 m from bulrush, with 15.7% of these locations occurring along the edge (0 m distance) of bulrush habitat, 29.2% occurring near bulrush (1–10 m), and 31.5% occurring ≥ 20 m from bulrush (overall mean distance = 16 m; Fig. 2). The highest proportion of vole captures in non-bulrush dominated habitats occurred during summer and early fall (May-September). During the same survey period, the one baited camera trap in non-bulrush-dominated habitat was placed in a Common Reed patch (100% cover), located >10 m from bulrush habitats. This camera was active for one six-week period during which voles were detected during the summer and fall seasons.

During the 2019–2020 occupancy survey period, there were 99 camera trap nights during the sampling period across six marshes, with voles being detected in six of the 14 camera traps stations (Table 1). Voles were detected at sites dominated by bulrush, Yerba Mansa, Boraxweed, rushes, and Common Reed; including at three locations (within Sites 9 and 17) where bulrush was completely absent (Table 1). It is of note that these detections, via cameras, do not indicate the number of voles detected, but simply the occurrence of voles outside of bulrush dominated habitats. In addition to camera detections, we observed multiple instances of vole sign on the periphery of bulrush patches of six of the seven sites surveyed for sign. We found vole sign in habitats dominated by rushes, Common Reed, Yerba Mansa, and Boraxweed.

Site specific vegetation descriptions.—*Site 5*: This site consisted of a moderately sized bulrush marsh adjacent to open water (Fig. 1). This bulrush patch was surrounded by saltgrass with small amounts Boraxweed. Voles live trapped in non-bulrush-dominated trap locations (n = 6) were captured in areas of > 15% saltgrass and < 5% bulrush (live and/or litter) cover, including three locations where bulrush (live and litter) was absent.

Site 8: This site consisted of moderately sized bulrush and cattail (*Typha* spp.) patches, centered along a stream and fed by multiple sources (Fig. 1). The bulrush patch was surrounded by patches of Yerba Mansa, saltgrass, Common Reed, Alkali Sacaton (*Sporobolus airoides*), mesquite (*Prosopis* spp.), and salt cedar (*Tamarix* spp.). While voles were detected at two camera locations, only one location was co-dominated by non-bulrush (Common Reed; Table 1). We did not find vole sign in the peripheral area of this site.

Site 9: This site consisted of a bulrush marsh surrounded by a well-developed margin of rushes, Yerba Mansa, Boraxweed, and saltgrass (Fig. 1). The site also included two substantial patches of Common Reed,
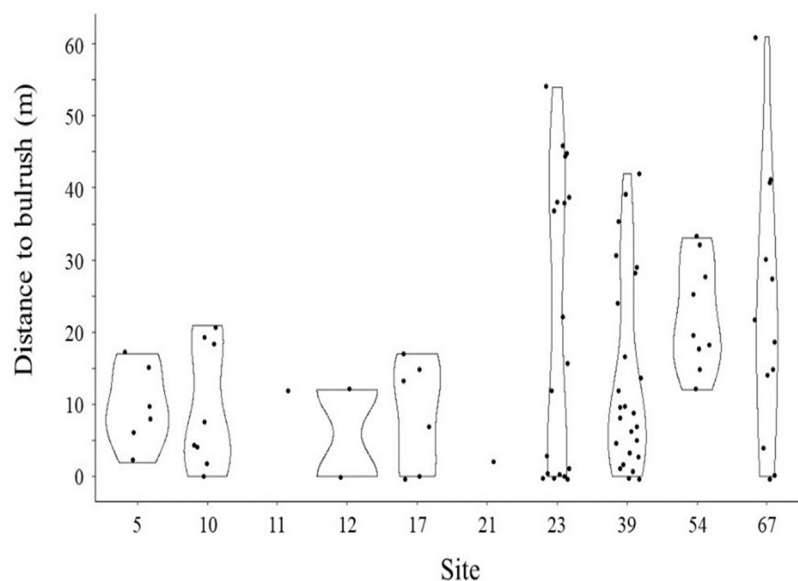

FIGURE 2. Violin plot showing the distance (m) of live trap locations that detected Amargosa Voles (*Microtus californicus scirpensis*) in non-bulrush dominated habitat to bulrush habitat within each sampled site during the 2015–2016 survey. Data collected from near Tecopa, Inyo County, California.

TABLE 1. Habitat notes for camera trap detections of Amargosa Voles (*Microtus californicus scirpensis*) in Sites 8, 9, 17, 22, and 58 in Tecopa, California, from 2019–2020. The abbreviation CN = camera identification number, SM/WD = soil moisture/water depth, and VD = voles detected (yes/no) with the number detected in parentheses.

Marsh	CN	Vegetation cover	SM/WD	VD	Notes
17	17.1	65% Yerba Mansa, 20% Boraxweed, litter depth 70cm	Moist soil	Yes (1)	10 trap/nights. A small patch of bulrush coming down hill. Vole sign, no burrow.
9	9.1	70% Boraxweed, 4% Common Reed, 2% Yerba Mansa, litter depth 40cm	Dry soil	Yes (3)	11 trap/nights. Vole sign present, burrow present; 20-25m from the edge of bulrush patch. Voles observed using the burrow a couple of times.
	9.2	40% rushes, 2% Boraxweed, other spp. 10%, litter depth~60cm	Dry soil	Yes (2)	9 trap/nights. Vole sign tunnel through the grass, burrow built in the Juncus.
	9.3	60% rushes, 5% Yerba Mansa, 5% Boraxweed, litter dept 70cm	Dry soil	No	5 trap/nights. bulrush edge at 40m to the camera trap. Poop signs and two burrows.
	9.4	70% rushes, 5% saltgrass, litter depth 55cm	Dry soil	No	5 trap/nights. ~30 pellets of poop vole. Burrow present.
	9.5	50% Common Reed, 5% Yerba Mansa, litter depth 20cm	Dry soil	No	5 trap/nights. Two burrows with vole signs.
22	22.1	75% Yerba Mansa, bulrush < 5%, woody debris 2%, litter depth 50cm	Dry soil	No	11 trap/nights; House mouse every day, no voles were recorded
8	8.1	60% Yerba Mansa, 30% rushes, 20% Boraxweed, litter depth ~55cm	Dry soil	No	9 trap/nights. No standing water. No vole signs observed.
8a	8a.1	85% Common Reed, 25% bulrush, litter depth ~65cm	Moist soil, near small stream	Yes (12)	4 trap/nights. Voles observed every day, up to 3 voles observed in single frame, one aggression event.
	8a.2	85% bulrush, 15% Common Reed, litter depth 75-100cm	25cm	No	4 trap/nights. No images captured
	8a.3	90% bulrush, 7% Common Reed, 3% cattail, litter depth up to 150cm	Litter too deep to determine	Yes (1)	4 trap/nights. One vole individual captured on 1/20 @ 9:30pm. One *Peromyscus* individual observed same day.
	8a.4	80% Common Reed, 15% bulrush, 10% cattail, litter 65-70cm deep	13cm	No	4 trap/nights. One possible observation of house mouse.
58	58.1	60% rushes, litter depth 50cm	Dry soil	No	9 trap/nights; Harvest mouse every day, no voles were recorded
	58.2	40% bulrush, 40% Yerba Mansa, 5% Boraxweed, litter depth ~60cm	Dry soil	Yes (3)	9 trap/nights. Burrow-like tunnel. Woody debris in the area. Vole signs.

uphill of the bulrush marsh. Outside of the bulrush marsh, voles were detected on camera at two locations dominated by Boraxweed and rushes, respectively (Table 1). Additionally, we found vole burrows into the soil layer and vole feces in non-bulrush habitats along the periphery of Site 9.

Site 10: This site consisted of a strip of bulrush following a stream that flowed from a culvert under a road (Fig. 1). The bulrush area was surrounded by Yerba Mansa, saltgrass, Boraxweed, sedges, rushes, Alkali Sacaton, Almutaster (*Aster pauciflorus*), and Goldenweed. We trapped voles in eight locations where no bulrush was present but at sedge (n = 1 site), Yerba Mansa-sedge co-dominant (n = 1), Yerba Mansa, (n = 1), Yerba Mansa-rush co-dominant (n = 1), saltgrass-Boraxweed co-dominant (n = 1), Rush (n = 2), rush-Boraxweed (n = 1) dominated trap locations.

Site 11: This site consisted of a relatively small to moderately sized bulrush area surrounded by saltgrass,

Boraxweed, rushes, Annual Sunflower, and Common Reed (Fig. 1). The site had no apparent water source other than seasonal upwelling of groundwater or perhaps a diffuse spring discharge. We trapped voles at one location outside of the bulrush area in Boraxweed dominated habitat.

Site 12: This site consisted of a moderately sized bulrush area adjacent to a seasonal pond and was surrounded by areas of Common Reed, saltgrass, Boraxweed, rushes, and upland vegetation (Fig. 1). We trapped voles at two non-bulrush dominated locations; one location completely lacked bulrush (live and litter) and the second location had minimal (< 0.5%) bulrush litter present. One location was dominated by saltgrass and the other by a Boraxweed-saltgrass mix.

Sites 17 and 21: These sites consisted of a large bulrush marsh surrounded by saltgrass wetlands on the north and west and upland habitat with some rushes and Boraxweed on the south and east side of the site (Fig.

1). Camera trapping detected a vole in a Yerba Mansa dominated habitat patch located adjacent to a stream on the hillslope above the site. We found vole sign (feces, clippings, burrows) along the entire waterway flowing from residences and a recreational vehicle park on the hill above Site 21. While some bulrush occurred in these uphill locations, they were often dominated by rushes and saltgrass with a few patches of Yerba Mansa. We trapped voles in these sites at seven locations where bulrush was not dominant, including four locations where bulrush was not present. These locations were dominated by saltgrass (n =5) or a saltgrass-rush mix (n = 2).

Site 22: This site consisted of a relatively small patch of bulrush mixed with Annual Sunflower and surrounded by Yerba Mansa, saltgrass, and Alkali Sacaton (Fig. 1). We found vole sign along the periphery of the bulrush area; however live-trapping and camera trapping did not detect voles outside of bulrush habitat at this site.

Site 23: This site was disconnected from other potential vole habitat patches by alkali desert playa and consisted of large bulrush patches adjacent to spring sources surrounded by a large Common Reed patch to the northeast and saltgrass and rushes along other portions of the site (Fig. 1). A camera trap detected voles within a 100% Common Reed patch. We captured voles at 19 trap locations located in non-bulrush dominated habitats, including 11 locations where bulrush was absent. These locations were dominated by saltgrass (n = 9), Boraxweed (n = 1), and Common Reed (n = 12) communities. Within the Common Reed patch, 10 locations lacked any bulrush presence.

Site 31: This site occurred in the extreme northern portion of the range of the species where voles had been translocated into restored desert wetland habitat (Fig. 1). The site consisted of bulrush areas along spring-fed streams and ponds, goldenrod (*Solidago* spp.) meadows, Common Reed patches, and upland areas dominated by mesquite and shrubs. Voles were only detected on camera in bulrush dominated habitat. While we observed most vole sign in bulrush areas, we found vole feces in mesic, marginal habitat around the periphery of bulrush areas.

Site 39: This site consisted of a central bulrush dominated area and was surrounded by saltgrass, rushes, and Yerba Mansa (Fig. 1). We trapped voles at 25 non-bulrush dominated trap locations, including 13 locations where bulrush was absent. These trap locations were in Rush (n = 7), saltgrass (n = 12), saltgrass-rush (n = 1), saltgrass-Boraxweed (n = 1), saltgrass-Goldenweed (n =1), rush-sunflower (n = 1), saltgrass-rush-Goldenweed (n = 1), saltgrass-rush-bulrush (n = 1), and Seaside Arrowgrass (*Triglochin concinna*, n = 1) dominated habitats.

Site 54: This site consisted of a large bulrush dominated area and was fed via a culvert by hot-spring water that originates at the head of Site 1 (Fig. 1). The bulrush area was surrounded by rushes and saltgrass. We captured voles at nine non-bulrush dominated trap locations, including three where bulrush was completely absent. These trap locations were located in saltgrass (n = 4), Rush (n = 3), and saltgrass-rush (n = 2) dominated habitats.

Site 58: This site consisted mostly of cattail dominated vegetation which followed a small stream flowing from a spring before entering a larger marsh area consisting of bulrush-cattail mixed habitat and which connected to other sites (Fig. 1). The site was surrounded by areas of relatively high plant diversity, with areas dominated by Yerba Mansa, rushes, Boraxweed and interspersed with mesquite and cottonwood (*Populus* spp.). Camera traps at this site detected voles at a location co-dominated by Yerba Mansa and relatively young bulrush (Table 1).

Site 67: This site consisted of two very small bulrush patches that were surrounded by patches of Yerba Mansa, Boraxweed, saltgrass, rushes, Alkali Sacaton, and Annual Sunflower (Fig. 1). We trapped voles at 12 trap locations where non-bulrush dominated trap locations, including 10 where bulrush was absent. These areas were in Rush (n = 2), Boraxweed (n = 2), Annual Sunflower (n = 2), saltgrass (n = 1), sedge (n = 1), rush-sunflower (n = 1), rush-Yerba Mansa (n = 1), Boraxweed-rush (n = 1), and saltgrass-Yerba Mansa-sunflower (n = 1) dominated areas.

DISCUSSION

While it is clear from previous works examining Amargosa Vole habitat use that voles are dependent on bulrush for their ecology (e.g., Klinger et al. 2016), we have shown non-bulrush dominated habitats are also used by the species, with up to 17% of trap locations in which we captured voles being located in non-bulrush dominated locations, especially habitats in Common Reed, rushes, sedges, and Boraxweed dominated communities. Vole habitat use and selection is complex may be driven by a variety of factors (Ostfeld et al. 1985; Lin and Batzli 2001; Yletyinen and Norrdahl 2008), and while outside the scope of this paper, we believe that the detection of Amargosa Voles in non-bulrush habitats is likely associated with interactions of local biological and ecological drivers. First, the use of non-bulrush habitats may be associated with dietary needs, as bulrush has low nutritional values and Amargosa Voles must consume other plant species, particularly species with higher protein content than bulrush, to meet basal metabolic and nutritional requirements (Castle et al. 2020a). These resources are most abundant in non-bulrush areas (Janet Foley et al., unpubl. report). With approximately 45% of trapping detections occurring within 10 m of bulrush, these detections may represent short distance excursions of voles into non-bulrush dominated habitats in search of needed forage resources. Castle et al. (2020a) noted that sedges, Beaked Spikerush (*Eleocharis rostrellata*), rushes, grasses (Poaceae), Yerba Mansa, Annual Sunflower, and saltgrass are important components of vole diets, and most of these plant

species were documented at our vole-detection locations. Beaked Spikerush and non-saltgrass grasses (e.g., Alkali Sacaton, *Sporobolus airoides*) were not dominant species at our vole-detection sites but have been detected within vole-occupied marshes and are often associated with the periphery of bulrush patches in this system (Rado and Rowlands 1984; Janet Foley et al., unpubl. report). Second, because Amargosa Voles are reliant on standing water in this system (Janet Foley et al., unpubl. report), and this likely partially explains their dependence on water-associated bulrush, voles may only be able to use non-bulrush areas when standing water is seasonally available (e.g., more standing water in summer; pers. obs.). Third, the use of non-bulrush areas may also be influenced by the population dynamics of the species. The majority of vole detections in non-bulrush habitat occurred in summer months, when the vole population is reaching the peak of its yearly cycle (McClenaghan and Montgomery 1998; López-Pérez et al. 2023), and our detections may indicate that carrying capacity has been reached within a site and voles are dispersing in search of adequate habitat (Lin and Batzli 2001) or due to factors such as competition, inbreeding avoidance, and mate searching (Le Galliard et al. 2012). These non-bulrush areas may represent important dispersal corridors between habitat patches. Whether Amargosa Voles can persist in these non-bulrush areas is unclear. We observed voles using burrows outside of bulrush habitats in Site 9 (Fig. 3), which may indicate continued use of non-bulrush habitat in this site, but no persistent populations of voles have previously been detected in non-bulrush habitats at other sites (Klinger et al. 2016; López-Pérez et al. 2019; Janet Foley et al., unpubl. report). Amargosa Voles most likely require bulrush patches for survival (Klinger et al. 2015, 2016), due to the insulative litter layer of bulrush providing protection against extreme temperatures and cover against predators, but further studies are needed to understand the complexity of range-wide habitat selection and subpopulation persistence for the species.

We detected more voles in non-bulrush habitat and generally at greater distances from bulrush habitat in the southern portion of the range of the species than the north. While this trend may have been caused by our sampling effort, there are also possible ecological explanations for this pattern. Sites in the north generally have larger bulrush patches and may allow for higher densities of voles to persist, lessening the need for dispersal to non-preferred habitats (Andreassen and Ims 2001). Southern sites tend to be more florally diverse and have more gradual transitions between vegetation communities, thus they may provide more opportunity for voles to use non-bulrush habitat. More research into specific causes of differences in habitat use between marshes may lead to greater insight into species biology and aid in the management of the species.

Despite its importance to vole survival, bulrush alone is not sufficient to support the species (Castle et al. 2020a) and non-bulrush habitats seem to also be important to vole ecology even though these areas have been underrepresented in the literature and management concern. We suggest that managers should manage both bulrush and non-bulrush areas as vole habitat. In particular, non-bulrush areas adjacent to or connecting bulrush habitats should be managed for their forage and as corridors for dispersal between core habitat patches. By ensuring adequate forage resources surrounding bulrush patches, managers may be able to positively influence vole biology and population viability (Jones 1990; Turchin and Batzli 2001; Forbes et al. 2014). By promoting non-bulrush vole habitat between bulrush patches, where bulrush is not adapted to local conditions (e.g., soil salinity, water availability), managers may be able create corridors between source populations in larger bulrush areas (Janet Foley et al., unpubl. report) and safeguard populations against deleterious effects associated with isolated populations. This could aid in populations re-establishing in sites following local extirpation. Doing so would support a functional
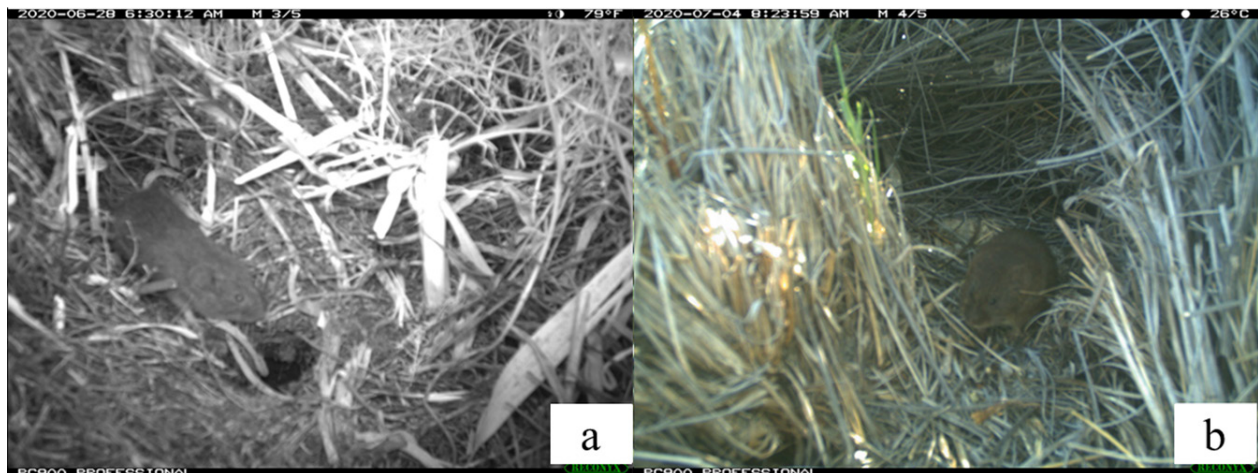

Figure 3. Images of Amargosa Voles (*Microtus californicus scirpensis*) captured using remote camera trapping techniques in 2019–2020 at Site 9 near Tecopa, Inyo County, California. Images depict (a) voles using below-ground burrows in Boraxweed (*Nitrophila occidentalis*) dominated habitat and (b) in Common Reed (*Phragmites australis*) dominated habitat.

metapopulation (Reed 2004; Molofsky and Ferdy 2005), which has been identified as necessary to the survival and recovery of the species (USFWS 2019; Castle et al. 2020b). Incorporating non-bulrush vole habitat into management objectives is likely to have multiple beneficial effects for the conservation of the vole as well as other rare and protected species in the area.

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Willow Flycatcher Use of Irrigation-Fed Wetlands in the Foothills of the Sierra Nevada of California

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Abstract.—Across California, Willow Flycatchers (*Empidonax traillii*) have been consistently and dramatically declining for decades. Fewer than 500 breeding pairs remain in the state despite the species being common and widespread less than a century ago. The species is extirpated from most of its historic range within California. Substantial conservation efforts have been made in the only known remaining strongholds of the species; mid-elevation Sierra Nevada meadows and riparian corridors of southern California. Previous reports, however, of Willow Flycatchers displaying territorial behaviors in irrigation-fed wetlands in low elevation areas of the Sierra Nevada foothills suggest additional areas may be either migratory stopover or breeding habitat. We followed up on prior anecdotal observations of Willow Flycatchers from the Sierra Nevada foothills, with the goal of confirming presence and meaningful use of the habitat. We detected flycatchers at nine of 14 wetlands surveyed and observed behaviors such as singing, counter-singing, and aggressive actions between individuals. Whether used as migratory habitat or breeding grounds, our observations demonstrate that Willow Flycatchers use irrigation-fed wetlands and continued management of these wetlands may be influential in the continued persistence of Willow Flycatchers in California.

Key Words.—breeding; *Empidonax traillii*; endangered species; small wetland; stopover

Introduction

As late as the 1940s, Willow Flycatchers (*Empidonax traillii*) bred across California, from sea level to around 2.400 m elevation and were commonly observed anywhere riparian scrub existed in the state (Grinnell and Miller 1944). Since then, the range of Willow Flycatchers in California has been reduced to only a few small pockets scattered across montane meadows of the northern Sierra Nevada and riparian corridors in southern California (Harris et al. 1987; Small 1994; Mathewson et al. 2013; U.S. Fish and Wildlife Service 2017), although it is a wide-spread and common migratory bird in North America (Sedgewick 2020). Where Willow Flycatchers persist in California, most populations continue to decline, with fewer than 600 breeding pairs remaining in the state (Loffland et al. 2014; U.S. Fish and Wildlife Service 2014). A few populations may be stable, however (Mary Whitfield, unpubl. data). The species is listed Endangered by California (California Department of Fish and Wildlife 2017). As such, all California populations are of conservation interest and identifying specific causes for their decline is critical.

Most historical declines are attributed to the widespread conversion of wetland habitat in California to agriculture (both cropland and wildland grazing) and urbanization (Sanders and Flett 1988; Frayer et al. 1989; Green et al. 2003). The reasons for ongoing declines in minimally disturbed and restored breeding habitat, however, especially in the Sierra Nevada where water resources are relatively abundant, remain unclear

(Loffland et al. 2022). Significant efforts have been taken to improve and restore breeding habitat for Willow Flycatchers in known populations, and there are now numerous meadows and waterways seemingly suitable for flycatchers that remain unoccupied (Schofield et al. 2018; Campos et al. 2020; Loffland et al. 2022). Despite ongoing declines in occupancy, when last monitored in the early 2000s those flycatchers remaining in the Sierra Nevada had breeding success and juvenile survivorship comparable to that of other related passerine species (Vormwald et al. 2011).

Like most migratory passerines, Willow Flycatchers spend most of their lives away from their breeding grounds (Lynn et al. 2003; Koronkiewicz et al. 2006), so improvements in the quality and availability of breeding habitat alone may not be sufficient to arrest or reverse population losses. Willow Flycatchers apparently have high wintering territory fidelity and relatively strong migratory connectivity linking breeding and wintering habitats (Koronkiewicz et al. 2006; Paxton et al. 2011; Ruegg et al. 2021; Mary Whitfield and Justin Shuetz, unpubl. report), suggesting that population trends in discrete breeding populations could be driven by effects in their wintering range or along migratory routes. Other studies have noted that many of the threats to Willow Flycatcher populations in the western U.S. are related to wintering and migration habitat (Paxton et al. 2017; Mary Whitfield and Justin Shuetz, unpubl. report), and Willow Flycatcher mortality is known to be at its highest during migration (Sillett and Holmes 2002; Paxton et al. 2017). Willow flycatchers have been found to maintain

low body fat stores during migration relative to other migratory passerines, suggesting they rely heavily on the presence of suitable stopover habitat to replenish depleted fat reserves (Yong and Finch 1997; 2002).

It is also possible declines in Willow Flycatcher populations in California are attributable to not just the availability of breeding habitat, but their ability to colonize that habitat. Although unoccupied breeding habitat is available in the Sierra Nevada and southern California, it is more sparsely distributed and widely dispersed than under historical conditions (Mathewson et al. 2013; U.S. Fish and Wildlife Service 2014) resulting in a metapopulation (Hanski 1998; Finch et al. 2002). The viability of the flycatcher metapopulation may be dependent on the degree of fragmentation and dispersion of habitat patches (Hanski 1998). Dispersal distances are relatively low in both adult (mean < 10 km) and juvenile (mean = 20.5 km) Willow Flycatchers in California (U.S. Fish and Wildlife Service 2014; Sedgewick 2020). Overall, current management plans suggest patches occur no more than 15 km apart to maintain connectivity within the metapopulation (Finch et al. 2002). In addition, Willow Flycatchers may rely on the presence of nearby conspecifics in selecting breeding territories which becomes less likely in a fragmented landscape and in the broader context of a declining metapopulation (Schofield et al. 2018).

Given that population declines are likely driven by pressures across all portions of their life cycle, conservation planning must consider the habitat needs of Willow Flycatchers in California across their breeding, wintering, and migration ranges, including migratory stopover sites. For this reason, we conducted systematic surveys to follow up on frequent anecdotal observations of Willow Flycatchers made in the Sierra Nevada foothills by avian surveyors with the University of California, Berkeley (UCB) between 2016 and 2019 (unpubl. data) who were performing targeted surveys of wetlands for California Black Rail (*Laterallus jamaicensis coturniculus*) and Virginia Rail (*Rallus limicola*). Records of Willow Flycatchers in this region from the UCB group have occurred during both the spring and fall migration seasons and the breeding season, with singing, counter-singing among multiple individuals, and interactions characteristic of territory defense. These behaviors are not necessarily indicative of breeding and no direct evidence of nest building or provisioning chicks have been documented, leaving the breeding status of these birds unknown. We systematically surveyed wetlands to assess whether these incidental observations represent a previously unknown breeding population or were evidence of an important migratory stopover site.

METHODS

Study area.—We conducted our study in the foothills of the Sierra Nevada in central California, focusing on small wetlands located at low elevations (< 1000 m) in Nevada, Yuba, and Butte counties (Fig. 1). The wetlands of the Sierra Nevada foothills are a hydrologically and
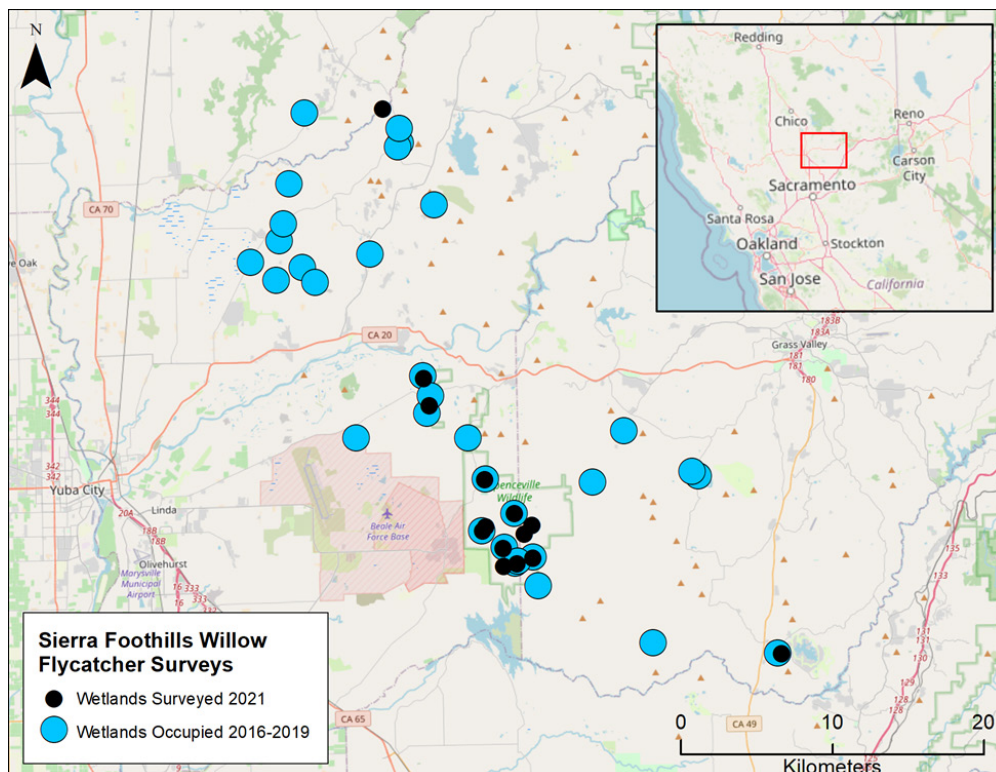

Figure 1. Wetlands with incidental Willow Flycatchers (*Empidonax traillii*) observations between 2016 and 2019 (blue dots) and sites where targeted Willow Flycatchers surveys were conducted in 2021 (black dots).

ecologically unique resource, fed largely by irrigation water from cattle ranching or other human activities (Huntsinger et al. 2017; Van Schmidt et al. 2021). Foothill wetlands are typically small (< 1 ha), and relatively sparsely distributed within a semi-arid matrix of oak savannah, open ranch land, and scattered development. Irrigation systems in the Sierra Nevada foothills are extensive and were initially established in the mid-1800s during the California gold rush (Van Schmidt et al. 2021). In contrast with much of the rest of California, the number of small wetlands has been increasing over time as more irrigation water is introduced to the landscape (Van Schmidt et al. 2021). Although the majority of wetlands in the study region are on private land, there are also many wetlands located on public land. For public wetlands, irrigation water is often explicitly designated for conservation efforts rather than commercial activities like ranching (Van Schmidt et al. 2021).

Field methods.—At the outset of this study, we compiled incidental Willow Flycatcher observations made by the UCB rail surveyors between 2016 and 2019 to help select wetlands for Willow Flycatcher surveys in 2021. Willow flycatcher records included both direct in-the-field observations made by UCB surveyors and recordings made using audio recording units. We only considered records that included the diagnostic the fitz-bew vocalizations characteristic of Willow Flycatchers as positive observations because it is difficult (or sometimes impossible) to differentiate this species from other closely related flycatchers by sight alone. UCB conducted rail surveys at between 225 and 275 wetland patches in the Sierra Foothills annually, 34 of which had positive Willow Flycatcher detections (Fig 1). Observations at six of these locations occurred during what is considered peak breeding season for Willow Flycatchers in the Sierra Nevada, between June 15 and July 15 (Bombay et al. 2003). Because these detections were incidental and not a part of standardized surveys, we did not incorporate these observations into statistical analyses.

Wetlands surveyed by UCB between 2016 and 2019 were selected for their suitability as Black and Virginia rail habitat, which is typified by open areas of rushes (*Juncus* spp.), cattails (*Typha* spp.), or sedge (*Carex* spp.). Wetland patches dominated by riparian shrubs, the habitat overwhelmingly favored by Willow Flycatchers, were not actively surveyed by UCB, and Willow Flycatcher observations made by UCB were typically at the periphery of their study areas. Because areas of riparian shrubs were generally not surveyed by UCB, in addition to targeting Willow Flycatcher surveys within appropriate habitat at wetlands with confirmed flycatcher observations, we also identified nearby wetlands that were not surveyed by UCB but had a high density of riparian shrubs suitable for flycatchers. We selected 14 priority wetlands and established survey points within those wetlands for Willow Flycatcher

surveys (Fig. 1). Survey points were located 50 m apart within wetland habitat and excluded from surrounding matrix habitat types. We selected 10 sites based on their proximity to previous flycatcher detections and four sites based on a qualitative assessment of habitat characteristics by experienced surveyors. Nine of the 11 wetlands previously surveyed by UCB had regular Black Rail detections. All survey sites were located on public land; 12 of the wetlands were within the Spenceville Wildlife Area owned and managed by the California Department of Fish and Wildlife (CDFW), one was along a public road, and one was at a water treatment plant.

We followed survey protocols described by Bombay et al. (2003) that use broadcasts of Willow Flycatcher vocalizations to elicit a response from nearby flycatchers. Surveys took place between 19 May and 17 July 2021; an interval that spans the majority of the breeding season within the Sierra Nevada region. We visited sites 1–9 times during that time span (mean = 3.3 visits), depending on detections, activity, and the assessment of an experienced surveyor on habitat suitability (Appendix Table). In California, both territorial and migratory individuals may be present between mid-May and mid-June (Bombay et al. 2003; Sogge et al. 2010). Depending on previous detections at a site, site characteristics, and subspecies, individuals displaying territorial behaviors between late-June and mid-July are more likely to be resident breeding birds (Bombay et al. 2003; Sogge et al. 2010). The phenology of Willow Flycatchers encountered in the foothills may be different from that of Willow Flycatchers that breed at mid and high elevations of the Sierra Nevada (Bombay et al. 2003), so we cannot confidently confirm breeding status based on the timing of observations alone.

In the nearest breeding habitat to the Sierra Nevada foothills, 15 June marks the point in which territorial singing decreases and is the initial date in which breeding status may be suspected in instances where previous observations were made during that season (Bombay et al. 2003). In the southern population, breeding status may first be suspected starting 1 June, assuming previous observations (Sogge et al. 2010). We used this timing to define a period between 15 June and 15 July that would indicate evidence that a wetland represents breeding habitat; however, due to the unknown phenology of these individuals, we did not consider any of these detections to be confirmation of breeding status. Breeding status would not be confidently inferred until a detection after 24 June in the southern population and 26 June in the Sierra Nevada population (Bombay et al. 2003; Sogge et al. 2010). Following the Sierra Nevada protocol, an individual detected between 15 and 25 June, but not detected after, would not necessarily be considered a migrant or otherwise absent because detection probability falls substantially due to reduction in singing rates post-25 June (Bombay et al. 2003).

Each day broadcast surveys began 30 min before local civil sunrise and continued until 1000. This interval maximizes detectability due to Willow Flycatcher activity. Upon arrival at a wetland, surveyors spent 10 min passively listening prior to beginning broadcast surveys. After this, surveyors spent 6 min at each pre-defined point alternately playing Willow Flycatchers vocalizations and listening for responses. Broadcast survey points covered the full extent of the available habitat within a wetland and were spaced approximately 50 m apart to maximize detection probability. We delineated broadcast points before the initiation of surveys based on satellite imagery, although we moved or added points during initial survey visits if needed to fully cover available habitat.

If one or more Willow Flycatchers were detected (or suspected but unconfirmed) during broadcast surveys, a follow-up survey was conducted either immediately after broadcast surveys or the following day. The goals of follow-up surveys were to: (1) relocate any birds detected; (2) confirm the bird species identification by listening for the characteristic vocalizations if necessary; (3) identify feeding perches, singing perches, and other areas of use; (4) watch for behaviors indicative of breeding such as carrying nesting material, carrying food or fecal sac, and interacting with possible mates; and (5) locate nests or fledglings if possible. If the breeding status of the individuals observed remained inconclusive, additional follow-up visits were conducted when possible. During both initial broadcast surveys and follow-up surveys, experienced surveyors recorded individual behavior and interactions. We considered non-agonistic interactions featuring quiet vocalizations characteristic of pair-bonding as evidence of opposite-sex pairs.

We characterized the dominant vegetation and hydrology at each wetland within the study once per season while consulting aerial imagery as described in Bombay et al. (2003). Site-scale vegetation data collected during surveys included overall percentage of the wetland covered with riparian deciduous shrub (RDS), rushes, forbs, and grasses, and percentage RDS comprised of willows (*Salix* spp.; Bombay et al. 2003). Surveyors also noted whether signs of American Beaver (*Castor canadensis*) were observed within the wetland and whether the water source was natural or not.

We compared percentage saturated soil, percentage cover RDS, percentaget cover grass, and percentage cover forbs of occupied versus unoccupied wetlands at sites surveyed for Willow Flycatchers using call-playback surveys using a Student's *t*-test and compared water source and apparent beaver presence using a Chi-square test. To meet parametric assumptions, we used the natural log transformation for percentage saturated soil, percentage cover RDS, and percentage cover forbs. Due to non-normality of data, we used a Mann-Whitney U-test to compare percentage cover of rushes and percentage RDS willow. Analytical methods such as Occupancy Models (Mackenzie et al. 2002) were inappropriate for this small dataset and would require further survey effort. We considered all results to be significant at a Bonferonni-corrected α-level of 0.00625. We report all values as the mean ± standard error.

RESULTS

We recorded 17 separate detections of Willow Flycatchers at nine of the 14 wetlands surveyed (64%), with singing observed at seven of these locations (50%;
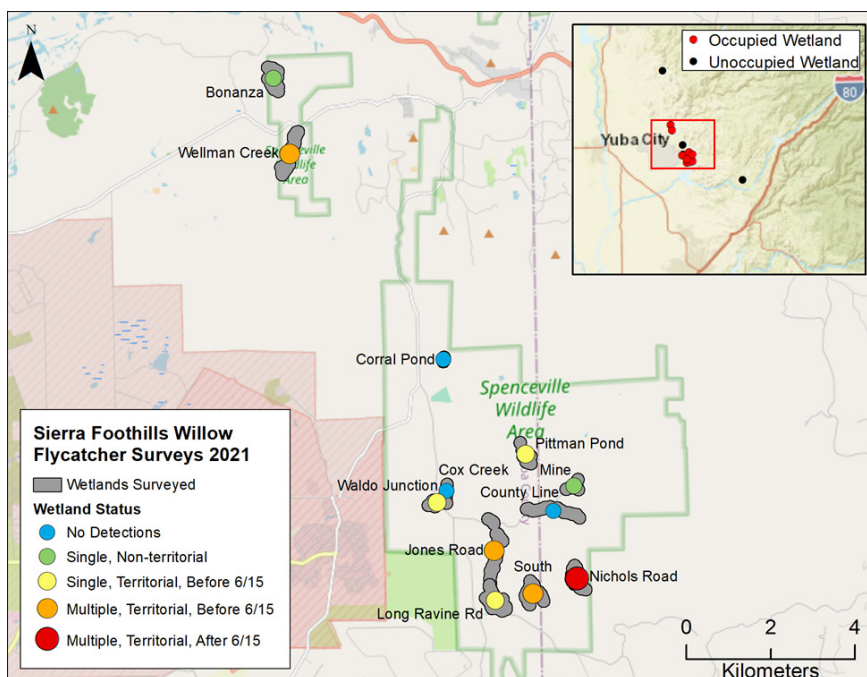

FIGURE 2. Location of wetlands in the Sierra Nevada foothills, including number, type, and timing of detections.

Fig. 2; Appendix Table). Five of the wetlands occupied by singing flycatchers (36%) had more than one individual detected on the same survey date and these individuals were observed interacting with one another (Appendix Table). The observations of Willow Flycatchers made at eight of nine occupied wetlands all took place prior to 15 June, the expected start of territoriality and breeding in nearby populations. One pair of Willow Flycatchers was observed on 15 June when Sierra Nevada Willow Flycatchers would be considered to be on their breeding territories, providing some evidence of a breeding attempt. This pair was observed singing and interacting with one another, which also typically indicates territoriality; however, there was no direct confirmation of a nest or young and no Willow Flycatchers were observed during surveys conducted after 15 June, so we could not conclusively ascertain breeding status.

In the majority of the wetlands where Willow Flycatchers were observed (n = 8), the flycatchers were detected on a single occasion. In the largest wetland in our study area (Wellman Creek), however, multiple individuals were observed interacting and displaying territorial behaviors during three separate survey visits. In early June the beaver dam that maintained the wetland broke, and the habitat quickly desiccated. After 12 June, Willow Flycatchers were no longer detected. Another of the wetlands where Willow Flycatchers were present (Bonanza) became dewatered early in the season due to the failing of the irrigation systems that typically feed the wetland. At the time of the initial visit to each wetland in early May, an average of 30.2% (± 4.88) of the total wetland area was inundated. In addition to these dewatering events, due to drought conditions in 2021; all wetlands were drier than they would be in typical years. The nearby Yuba River Marysville USGS water gauge recorded a mean annual flow rate in 2021 that was 23% of the historical 1970–2000 average annual flow (U.S. Geological Survey 2022). Habitat characteristics were similar between the nine occupied and five unoccupied wetlands (Table 1).

DISCUSSION

Our systematic surveys confirmed Willow Flycatcher presence in nine of 13 surveyed low-elevation wetlands in the Sierra Nevada foothills during late spring and early summer, demonstrating that the wetlands we surveyed are at a minimum extensively used by Willow Flycatchers during the migratory phase of their life cycle. Our observations also provide evidence (though not conclusive proof) that the wetlands surveyed may also represent breeding habitat. Further investigation will be necessary to identify the extent to which Willow Flycatchers use and rely on wetlands in the Sierra Nevada foothills, but we have confirmed that wetlands in the Sierra Nevada foothills provide resources to this species, whether used for breeding and migration or migration alone. The timing of the presence of Willow Flycatchers in our study suggests that these wetlands may serve as a stopover site for populations breeding either at higher elevation regions of the Sierra Nevada or further north. The mean arrival date for flycatchers detected in the mid and high-elevation Sierra Nevada (Bombay et al. 2003) and northern populations (http://www.ebird.org) corresponded to reduced detections at our foothill sites. Genetic sampling or tracking efforts may make it possible to identify which breeding population these individuals come from and could even provide evidence as to whether they represent a distinct population. There may be limited ability to differentiate breeding populations of Willow Flycatchers in western North America, however (Ruegg et al. 2021).

Managing wetlands to meet Willow Flycatcher needs in the Sierra Nevada foothills could be beneficial to Willow Flycatchers regardless of their breeding status, as both breeding and migratory stopover require similar habitat (Sedgewick 2020). Willow flycatchers historically nested in this region (Grinnell and Miller 1944), and with proper management preventing dewatering until after the breeding season, foothill habitats might once again provide opportunities for dispersing Willow Flycatchers

TABLE 1. Statistical tests comparing habitat characteristics of wetlands occupied by Willow Flycatchers (*Empidonax traillii*) with those not occupied by Willow Flycatchers. Significance based on a Bonferonni-corrected α-level of 0.00625. The abbreviation RDS = riparian deciduous shrubs and df = degrees of freedom. Numbers are means ± standard error.

Habitat Covariate	x̄ Occupied	x̄ Unoccupied	Test Statistic	df	*P*-value
% Saturated Soil	26.7 (± 8.53)	38.0 (± 14.23)	$t = 1.12$	11	0.298
% Cover RDS	31.1 (± 7.16)	30.5 (± 10.08)	$t = -0.24$	11	0.818
% Cover Grass	29.8 (± 3.24)	32.0 (± 5.65)	$t = 0.64$	11	0.733
% Cover Juncus	14.6 (± 2.75)	3.0 (± 2.68)	$U = 3.50$	—	0.030
% Cover Forbs	8.7 (± 2.90)	14.8 (± 5.74)	$t = 0.56$	11	0.600
% RDS Willow	71.6 (± 10.7)	75 (± 14.8)	$U = 22.0$	—	0.587
Beaver Presence	3/9	1/4	$X^2 = 0.09$	1	0.764
Natural Water Source	5/9	3/4	$X^2 = 0.44$	1	0.506

originating in other portions of its range. Although the Sierra Foothills are hotter and drier than the area currently inhabited by the Sierra Nevada population, Willow Flycatchers have the evolutionary potential to use and adapt to hotter climates if other habitat parameters are suitable (Forester et al. 2023; Schofield et al. 2023).

Currently, the wetlands in our study area are primarily managed to support rail species, especially the Black Rail which, unlike the shrub-associated Willow Flycatcher, requires open habitat dominated by sedges and rushes (Richmond et al. 2010). A management approach that supports both species would also be beneficial to other meadow-associated animals. Historically, wetlands in the Sierra Foothills typically contain a mix of cover types (van Schmidt et al. 2021) and encouraging that heterogeneity could help promote wetland ecosystem health as a whole. Water availability is the most important factor in maintaining appropriate habitat for both Black Rails (Richmond et al. 2010) and Willow Flycatchers (Mathewson et al. 2013). Artificial irrigation is a primary strategy for providing water to wetlands in this region (Huntsinger et al. 2017; Van Schmidt et al. 2021), mimics historic conditions, and could benefit both species. Although most management and restoration activities have occurred on public land, water use and management in California is complex and involves many different stakeholders on private land and industry that have differing needs for water (Huntsinger et al. 2017; Van Schmidt et al. 2021). Another strategy for maintaining water on the landscape with or without supplemental irrigation could be encouraging beaver presence, which would benefit both Black Rails and Willow Flycatchers by retaining water later into the dry season. As we observed in the Wellman Creek site, the wetland became rapidly dewatered and transitioned to unsuitable habitat after the loss of a beaver dam.

The drought conditions experienced across California in 2021 (Seager et al. 2022), combined with the abrupt loss of existing water at the locations maintained by beaver dams and irrigation, confound the interpretation of our observations. It is possible that these conditions prevented breeding or resulted in nest failures in locations that commonly support breeding Willow Flycatchers during more favorable years. Further investigation is needed to determine whether Willow Flycatchers use the Sierra foothills for breeding habitat and to what extent wetlands in the foothills are used during migration. Historically, the avifauna of the Sierra foothills has been relatively poorly studied; the extensive breeding population of Black Rails, for example, was not discovered until 1994 (Girard et al. 2010). We can reasonably suspect that if Willow Flycatchers were breeding in the central Sierra foothills, their phenology would be different from that of flycatchers nesting in the mid to high-elevation Sierra Nevada, where green-up is significantly later, and likely more similar to populations nesting in southern California (e.g., Kern River Valley).

Although formerly characterized by abundant wetland habitat, the Central Valley of California and the adjacent low-elevation portions of the Sierra Nevada foothills lost 86% of historical wetlands between 1936 and 1989 (Frayer et al. 1989). Protecting and maintaining remnant wetlands is thus critical for species dependent on these habitats for migration and reproduction.

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Appendix Table. Survey dates and observations made during Willow Flycatcher surveys conducted in the Sierra Nevada foothills in 2021.

Site Name	Survey Date	Number of Willow Flycatcher Detected	Behaviors/Notes
Bonanza	May 19	0	
Bonanza	June 6	1	Foraging
Bonanza	June 24	0	
Bonanza	July 12	0	
Corral	June 2	0	
County Line	May 23	0	
County Line	June 22	0	
Cox Creek	July 6	0	

APPENDIX TABLE (CONTINUED). Survey dates and observations made during Willow Flycatcher surveys conducted in the Sierra Nevada foothills in 2021.

Site Name	Survey Date	Number of Willow Flycatcher Detected	Behaviors/Notes
Honcut Creek	June 10	0	
Honcut Creek	June 20	0	
Jones Road	May 27	1	Singing Male
Jones Road	June 29	0	
Jones Road	July 8	0	
Jones Road	July 15	0	
Lake of the Pines	June 11	0	
Long Ravine	June 13	1	Singing Male
Long Ravine	May 25	0	
Mine	June 3	1	Singing Male
Mine	June 8	0	
Mine	July 17	0	
Nichols Road	June 5	0	
Nichols Road	June 15	1	Singing Male + Female
Nichols Road	June 26	0	
Nichols Road	July 7	0	
Pittman Pond	June 3	1	Singing Male
Pittman Pond	June 8	0	
Pittman Pond	July 1	0	
Pittman Pond	July 14	0	
South Site	May 28	1	Singing Male
South Site	June 19	0	
South Site	July 2	0	
South Site	July 15	0	
Waldo Junction	May 25	1	Singing Male
Waldo Junction	May 29	0	
Waldo Junction	June 16	0	
Waldo Junction	July 6	0	
Waldo Junction	July 18	0	
Wellman Creek	May 20	1	
Wellman Creek	May 24	1	
Wellman Creek	May 29	1	
Wellman Creek	June 1	2	Counter Singing Males
Wellman Creek	June 2	3	Counter Singing Males
Wellman Creek	June 12	1	Beaver Dam Broke
Wellman Creek	June 23	0	
Wellman Creek	July 3	0	
Wellman Creek	July 18	0	

Long-term Variation in Breeding Populations of Colonially Nesting Cormorants and Herons in a Severely Impaired Ecosystem at Clear Lake, California

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Abstract.—Clear Lake, a large and shallow lake in Lake County, northern California, USA, is highly eutrophic and severely impaired by human activities. We studied populations of colonially breeding Double-crested Cormorants (*Nannopterum auritum*), Great Blue Herons (*Ardea herodias*), Great Egrets (*Ardea alba*), and Black-crowned Night-Herons (*Nycticorax nycticorax*) by counting nests at Clear Lake and associated wetlands during 2006–2024 and compiled historical data (1993–2000). We found 10 colony sites, some subsequently abandoned and others recently colonized. Breeding populations of Double-crested Cormorant and Great Blue Heron declined during 1993–2024, but populations of the former increased and the latter were stable during 2011–2024. Populations of Great Egret and Black-crowned Night-Heron were stable during 2009–2024. The number of nests for all species was unaffected by water level. The causes of long-term population declines are unknown. Future monitoring is needed to better understand long-term trends and the environmental drivers of change.

Key Words.—Ardeidae; *Ardea alba*; *Ardea herodias*; coloniality; nesting; *Nannopterum auritum*; *Nycticorax nycticorax*; Phalacrocoracidae

Introduction

Located in the coastal ranges of northern California, USA, at an elevation of 402 m above sea level, Clear Lake (38°56′46″ to 39°07′23″N, 122°38′04″ to 122°54′46″W), in Lake County, is considered the oldest natural freshwater lake in North America (Sims 1988). Although relatively large with a surface area of 176.7 km² and 114 km of shoreline, it is relatively shallow, averaging 8.1 m deep with a maximum depth of 18.4 m (Horne and Goldman 1972). Because of its shallow depth, Clear Lake represents a polymictic and highly eutrophic lacustrine ecosystem with an abundance of nutrients that nourish cyanobacterial and algal blooms, especially during the warm summer months (Goldman and Wetzel 1963; Richerson et al. 1994; Winder et al. 2010), and an abundance of zooplankton and higher trophic level organisms, including many species of fish (Thompson et al. 2014) and waterbirds (Cooper 2004).

Clear Lake and its adjacent wetlands are severely impaired by a long history of human activities, including contamination of mercury (Hg) from a nearby mine, invasive species of aquatic plants and fishes, applications of herbicides and pesticides to control plant and animal pests, loss of wetlands due to modification and reclamation for agriculture and urban development, and cultural eutrophication from excessive nutrient loading (see reviews by Richerson et al. 2000, 2008; Suchanek et al. 2003; Thomson et al. 2013; Smith et al. 2023). In 1949, 1954, and 1957, massive amounts of the organochlorine pesticide dichlorodiphenyldichloroethane (DDD) were dumped in the lake in an attempt to control aquatic larvae of the Clear Lake Gnat (*Chaoborus astictopus*; Hunt and Bischoff 1960). Shortly after the second and third applications, large numbers of dead Western Grebe (*Aechmophorus occidentalis*) and Clark's Grebe (*A. clarkii*) were found along the shoreline. Subsequent studies revealed elevated concentrations of DDD in the tissues of several fishes, frogs, and the piscivorous grebes, providing the first documented instance of biomagnification in which toxic chemicals accumulated in increasingly higher concentrations from lower to higher trophic level organisms (Hunt and Bischoff 1960; Carson 1962; Rudd 1964).

Although the devastating effects of DDD on the grebes of Clear Lake are well documented, resulting in mass mortality of adults and cessation of breeding followed by a gradual multidecadal recovery of their breeding populations (Hayes et al. 2022), the impact of DDD on other piscivorous birds, including cormorants and herons, has not been determined. In March 1895, Chamberlin (1895) described a breeding colony of about 100 Double-crested Cormorant (*Nannopterum auritum*) nests in the vicinity of Reeves Point and a second immense colony stretching across what he estimated as half a mile of shoreline south of The Narrows, providing the only data on its breeding population prior to the application of DDD. No information is available on pre-DDD populations of colonially breeding herons. After describing the demise of the grebes of Clear

Lake following the application of DDD, Rudd (1964) thought that populations of egrets and herons also were lower than in former years but did not provide any data. The first post-DDD surveys of breeding populations of cormorants and herons at Clear Lake occurred during 1993–1994 as part of a study of biomagnification of DDD and Hg in birds and mammals (Wolfe and Norman 1998). Additional surveys were conducted during 1995–1996, 1998–2000 (for Double-crested Cormorant only during 1995–1996 and 1998–2000), and 2009–2012 (the latter surveys with data we supplied; Shuford 2010, 2014; Shuford et al. 2020a,b), and the nesting of Great Blue Heron in different colonies was briefly summarized by Lyons (2023). These surveys revealed large numbers of breeding Double-crested Cormorants, Great Blue Herons (*Ardea herodias*), and Black-crowned Night-Herons (*Nycticorax nycticorax*), and smaller numbers of Great Egrets (*Ardea alba*).

Given the potential threats of habitat loss, water diversions, introduced species, bioaccumulation and biomagnification of toxic chemicals, climate change, and other threats to waterbirds, monitoring their populations and habitats is crucial for evaluating their conservation status, detecting long-term population trends, assessing habitat quality, and documenting the effects of environmental changes and management practices on waterbirds (Kushlan et al. 2002). We summarize long-term variation in breeding populations of cormorants and herons at Clear Lake based on data from previously published surveys during 1993–2000 and new surveys during 2006–2024. We discuss the potential causes of variation and provide suggestions for future monitoring.

Methods

Study area.—Clear Lake has three major arms (Fig. 1). Major wetlands occur along large tributaries at the northwest end of the lake (Middle Creek and Rodman Slough), at its outlet at the southeast end of the lake (Anderson Marsh and Cache Creek), and in many shallow areas along the margins of the lake, especially at the mouths of small tributaries (Fig. 1). Riparian forests dominated by willows (*Salix* spp.), Valley Oak (*Quercus lobata*), Fremont Cottonwood (*Populus fremontii*), and California Sycamore (*Platanus racemosa*) occur along the shores of tributaries and portions of the lake, providing nesting habitat for cormorants and herons. Urban development occurs along the shore in several areas of the lake, but not at Rodman Slough, Anderson Marsh, and most of Cache Creek. Recreational boating and fishing are extremely popular, with dozens or hundreds of watercraft on the lake daily during fair weather.

Precipitation occurs mostly during October to May and rarely during June to September (Suchanek et al. 2008), with the water level typically highest during February to April, declining during summer, and lowest October to December (De Leon and Deligiannis 2022).
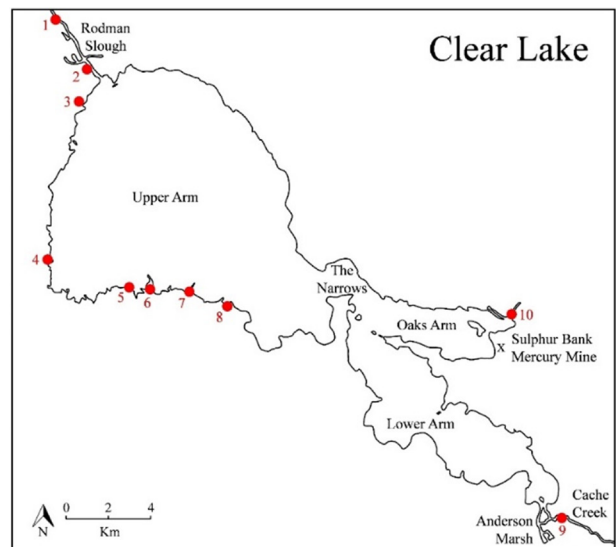

Figure 1. Map of Clear Lake, California, and associated wetlands with locations of 10 colony sites of breeding cormorants and herons indicated by red circles. See Table 1 for the number, name, and characteristics of each colony site.

The level of Clear Lake is measured in reference to the Rumsey gauge, which was established by Captain Rumsey at Lakeport in 1873. Zero Rumsey is considered the natural low water level of Clear Lake. Zero Rumsey is equal to 401.805 m (1,318.257 ft) above mean sea level. A full lake, by definition, is reached when the lake measures 2.30 m (7.56 ft) on the Rumsey gauge (https://www.lakecountyca.gov/DocumentCenter/View/4336/Historical-High-and-Low-Water-Levels-of-Clear-Lake-PDF). Water level at Lakeport (west shore of Upper Arm) varied dramatically since monitoring of breeding cormorants and herons began in 1993, ranging from 3.22 m (10.58 ft) Rumsey in 2017 to -0.835 m (-2.74 ft) Rumsey in 2022 (Fig. 2), but well within the historical extremes of -1.07 m (-3.50 ft) Rumsey in 1920 and 4.16 m (13.66 ft) Rumsey in 1890 (Suchanek et al. 2003; De Leon and Deligiannis 2022).

Breeding surveys.—During 2006 to 2024, we intermittently searched for breeding colonies of cormorants and herons along the shores of Clear Lake, adjacent tributaries, and the outlet. We did not search any of the colony sites annually, with some searched during more years than others, and some searched up to 10 times within a breeding season. We made visual surveys from a canoe, motorboat, or from land during the breeding season, from 1 January to 14 June, but mostly in late March and throughout April. During each survey we counted or estimated the number of active nests as a measure of reproductive effort (no data were obtained on reproductive success, such as the number of nestlings), and we identified the species that constructed and attended each nest, usually with the aid of binoculars or a telescope. Obtaining an exact count was often difficult due to nests hidden in the foliage and branches of trees,
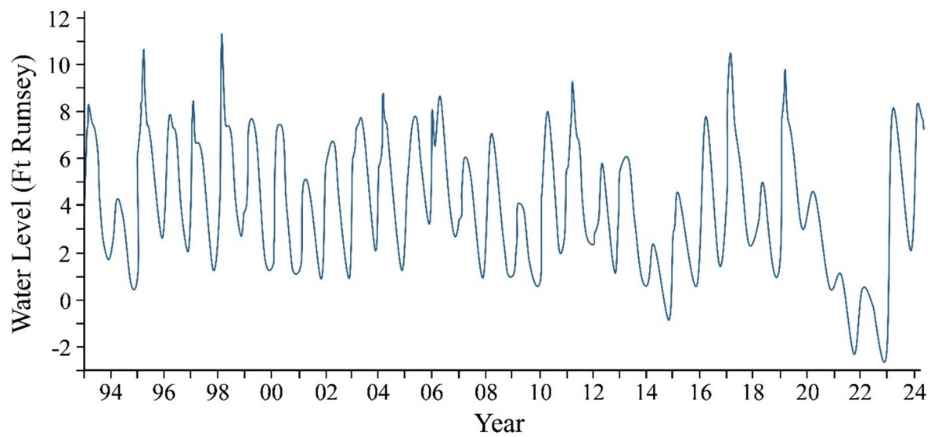
FIGURE 2. Water level at Lakeport, Clear Lake, California, from January 1993 through May 2024. Water level at Clear Lake has historically been measured as number of feet Rumsey (see Methods for definition of this measurement).

especially during April to June. For sites surveyed multiple times, we used the count with the highest number of nests as our measure of reproductive effort. We considered clusters of nests with gaps of less than 1 km from the nearest cluster of nests sub colonies of a single colony. Priority in naming colony sites was given to established names for prominent topographic features rather than urban developments (the latter are more likely to change over time). We obtained coordinates of the approximate center of each colony site from Google Earth (http://www.google.com/earth).

Statistical analyses.—We obtained water level data during the study period from the U.S. Geological Survey (http://waterdata.usgs.gov/ca/nwis/uv?site_no=11450000). To examine the relationship between water level and reproductive effort, we used the water level (ft Rumsey) on 15 March (early in the breeding season, when many birds were still deciding when and where to nest) for each year and the total number of nests in all colonies combined during years in which the major colonies of each species were all monitored in Rodman Slough, the Upper Arm, and Cache Creek, including previously published data from 1993–2000. We used Linear Regression (Zar 2010) to regress the number of nests against the independent variables year and water level, separately, for each species (sample sizes were too small to use multiple regression). We used Statistix 10 software (Analytical Software, Tallahassee, Florida) for all descriptive statistics and inferential statistical tests with an α = 0.05.

RESULTS

Colony site dynamics.—Colonially breeding cormorants and herons nested in 10 distinct colony sites: two exclusively in the largest tributary of the lake (Upper Rodman Slough and Lower Rodman Slough), seven along the shores of the lake or nearby in small tributaries, and one in the outlet of the lake, Cache Creek (Fig. 1,

Table 1). Four colony sites (Reeves Point, Quercus Point, Cache Creek, and Clearlake Oaks Wetlands) hosted all four study species; the other six sites hosted one to three species (Fig. 3, Tables 1–2). Several colony sites had distinct sub colonies separated by gaps of several hundred meters, including Willow Point (Library Park and Willow Point), Reeves Point (Reeves Point and mouth of Adobe Creek), Long Tule Point (McGaugh Slough and Shirley Slough), and Cache Creek (west and east sections).

Excluding the years 1995–2000, when Shuford (2010, 2014) surveyed some colony sites for cormorants but not herons, two colony sites, Long Tule Point and Cache Creek, were active each year surveyed for 14 y and one site, Quercus Point, was active each year surveyed for 10 y. Two colony sites, Upper Rodman Slough and Lower Rodman Slough, were abandoned without being reused during the study period. Upper Rodman Slough was active each year (although one year was not monitored) from 2011–2019, but no nests were found in 2024, and Lower Rodman Slough was active each year during 2011–2014 but not during 2016–2019 or 2024. Two colony sites, Willow Point and Reeves Point, were intermittently active and inactive. Three recently discovered colony sites were either previously overlooked or represented new colonizations. We detected the Lyons Creek colony site in 2024; if not previously overlooked, it may have been established by Great Blue Herons previously nesting 3.4 km away at Upper Rodman Slough. The Kelsey Creek colony site was first colonized by a single pair of Great Blue Herons in 2021 in an area frequented by birders (including ourselves) who had not seen it during the previous 15 y. We first detected nests at Clearlake Oaks Wetlands in 2020, but we may have previously overlooked these nests. The latter two colony sites remained active annually with an increasing number of nests through 2024.

Species accounts.—The Double-crested Cormorant nested at six colony sites (Table 2) with a maximum

TABLE 1. Colony sites for breeding cormorants and herons at Clear Lake and associated wetlands, including site number corresponding with Figure 1, coordinates (decimal degrees north, west), years surveyed, years active, and species nesting (BCNH = Black-crowned Night-Heron, *Nycticorax nycticorax*; DCCO = Double-crested Cormorant, *Nannopterum auritum*; GBHE = Great Blue Heron, *Ardea herodias*; GREG = Great Egret, *Ardea alba*). Notes and references are a - surveys reported by Wolfe and Norman (1998), b - referred to as Library Park (Shuford 2014), c - surveys reported by Shuford (2014) and Shuford et al. (2020b), referred to as Mouth of Holiday Cove, d - surveys reported by Shuford (2014), e - referred to as east of Quercus Point for Double-crested Cormorant and west of Clear Lake State Park for Great Blue Heron (Shuford 2014, Shuford et al. 2020b), f - surveys reported by Wolfe and Norman (1998), referred to as Slater Island, and g - surveys reported by Shuford (2014) and Shuford et al. (2020b), referred to as Slater Island, Anderson Marsh.

Colony	Coordinates	Years surveyed	Years active	Species
1. Upper Rodman Slough	39.1372, -122.9025	1993–1994a, 2011–2014, 2016–2019, 2024	1993–1994, 2011–2014, 2016–2019	DCCO, GBHE, GREG
2. Lower Rodman Slough	39.1214, -122.8914	2011–2014, 2016–2019, 2024	2011–2014	GREG
3. Lyons Creek	39.1058, -122.8974	2024	2024	GBHE
4. Willow Point	39.0425, -122.9136	2009–2019b, 2024	2009–2016, 2024	BCNH
5. Reeves Point	39.0275, -122.8794	1998-2000c, 2006, 2008–2012, 2014, 2018–2019, 2024	1999–2000, 2006, 2008–2009, 2018–2019, 2024	BCNH, DCCO, GBHE, GREG
6. Long Tule Point	39.0258, -122.8577	1999d, 2006, 2008–2019, 2024	2006, 2008–2019, 2024	DCCO, GBHE
7. Quercus Point	39.0275, -122.8378	1993–1996e, 1998–2000e, 2006, 2008–2011e, 2024	1993–1998, 2000, 2008–2011	BCNH, DCCO, GBHE, GREG
8. Kelsey Creek	39.0203, -122.8158	2006–2024	2021–2024	GBHE
9. Cache Creek	38.9325, -122.6342	1993–1994f, 1999g, 2009–2019g, 2024	1993–1994, 1999, 2009–2019, 2024	BCNH, DCCO, GBHE, GREG
10. Clearlake Oaks Wetlands	39.0175, -122.6611	2020–2024	2020–2024	BCNH, DCCO, GBHE, GREG

count at a single site of 210 nests at Quercus Point during 1993 or 1994 (year not specified; Wolfe and Norman 1998), where we never detected nesting during 2006–2024. Our highest count at a single site was 100 nests at Reeves Point in 2006. Nesting activity began as early as 1 January and peaked in April (Appendix). Our maximum annual count for the lake was 101 nests in four sites in 2024, considerably lower than the 375 nests at three colony sites in 2000 (Shuford 2010). There was a significant negative relationship in the annual number of nests at all major colony sites combined from 1994–2024 (slope = -6.6, $F_{1,12}$ = 10.75, P = 0.007, r^2 = 0.47, n = 14 y of data). The number of nests at all major colony sites combined for this species increased significantly for our surveys from 2011–2024 (slope = 5.5, $F_{1,7}$ = 12.48, P = 0.010, r^2 = 0.64, n = 9 y; Fig. 4).

The Great Blue Heron nested at eight colony sites (Table 2) with a maximum count at a single site of 121 nests at Upper Rodman Slough in 1993 (Wolfe and Norman 1998), surpassing our highest count of 85 nests at Long Tule Point in 2010. Nesting activity began as early as 6 January and peaked in April (Appendix). The maximum annual count for the lake was 286 nests in three colony sites in 1992 (Wolfe and Norman 1998), exceeding our highest count of 191 in three sites in 2012. The annual number of nests at all major colony sites combined decreased significantly from 1993–2024 (slope = -6.3, $F_{1,9}$ = 15.73, P = 0.003, r^2 = 0.64, n = 11 y), but there was no significant relationship during our surveys from 2011–2024 ($F_{1,7}$ = 2.89, P = 0.133; n = 9 y; Fig. 4).

The Great Egret nested at six colony sites (Table 2) with a maximum count at a single site of 20 nests at Cache Creek and our maximum annual count for the lake of 27 nests at three colonies in 2018. Nesting activity began as early as 1 March and peaked in May (Appendix). Despite reporting large numbers of Great Blue Heron nests, Wolfe and Norman (1998) did not report this species nesting at Clear Lake, but it is unclear if they were absent or unreported. The annual number of nests at all major colony sites combined from 2011–2024 did not change significantly ($F_{1,7}$ = 1.88, P = 0.213; n = 9 y; Fig. 4).

The Black-crowned Night-Heron nested at five colony sites (Table 2), with a maximum count at a single site of 102 nests at Willow Point in 2016 and a maximum annual count for the lake of 109 nests at two sites in 2024. Nesting occurred during April to June, peaking later than other heron species (Appendix). Wolfe and Norman (1998) reported only three nests at Quercus Point in 1993 or 1994 (year not specified), where we never detected nesting, but they may not have visited Willow Point. This species was the only heron nesting at Willow Point, with annual counts of 32–104 nests during 2009–2016, mostly at Library Park with smaller numbers within three city blocks to the west and at a campground just south of the park. No nests were detected during 2017–2019, however, when the loud cries of a raptor were broadcast from loudspeakers in trees at Library Park to discourage herons from nesting. We did not survey the colony site during 2020–2023, but of 95 nests we counted in 2024, only one was in Library Park, where no raptor calls were broadcast, while all others were about 150 m to the
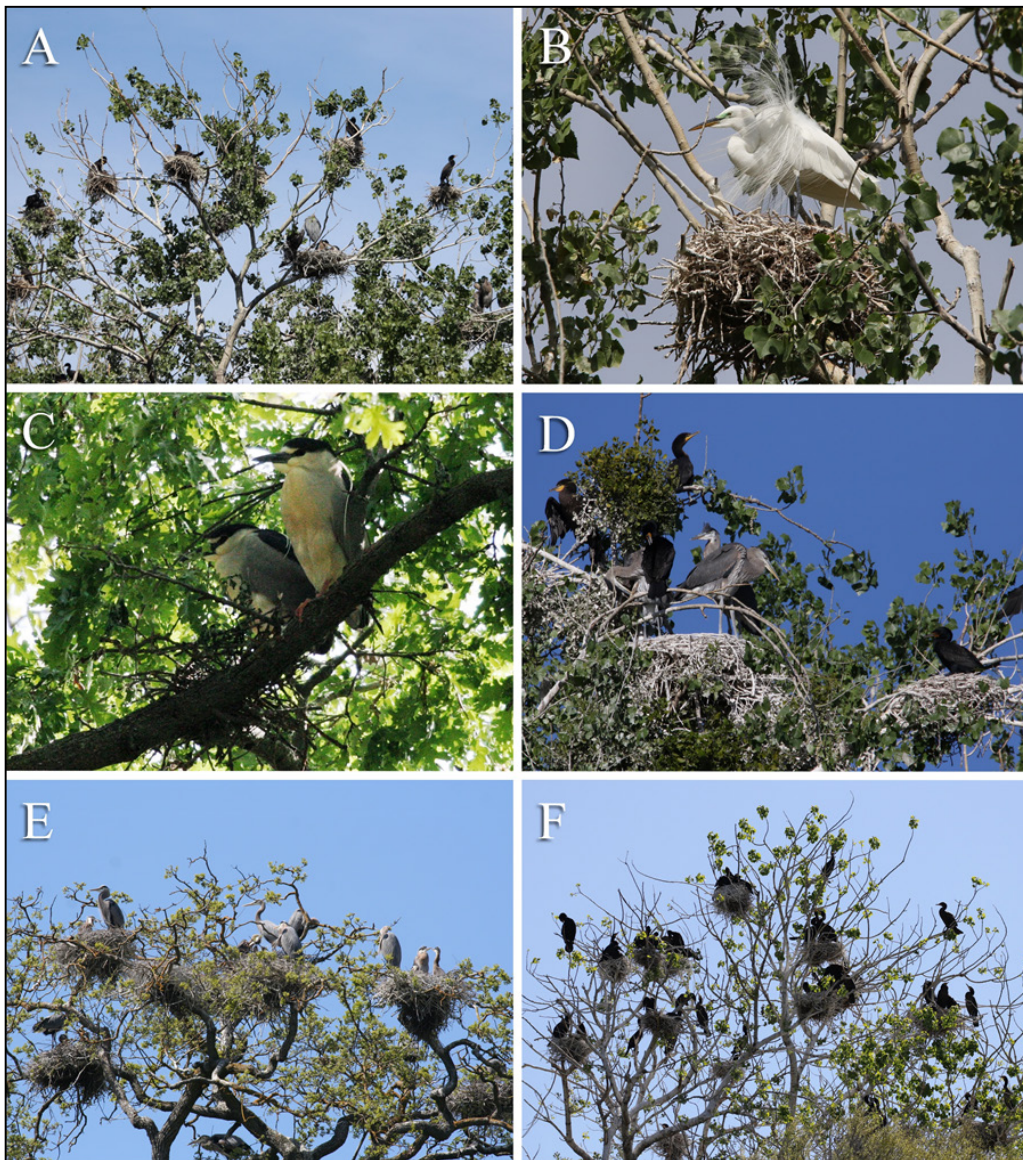
Figure 3. Bird species (BCNH = Black-crowned Night-Heron, *Nycticorax nycticorax*; DCCO = Double-crested Cormorant, *Nannopterum auritum*; GBHE = Great Blue Heron, *Ardea herodias*; GREG = Great Egret, *Ardea alba*) nesting at different colony sites at Clear Lake, California, and associated wetlands. **(A)** DCCO and GBHE at Upper Rodman Slough, 24 April 2016. **(B)** GREG at Upper Rodman Slough, 30 May 2017. **(C)** BCNH at Willow Point, 5 May 2012. **(D)** DCCO and GBHE at Reeves Point, 12 May 2017. **(E)** DCCO and GBHE at Long Tule Point, 24 April 2010; **(F)** DCCO at Clearlake Oaks Wetlands, 11 April 2024. (A, C, E, F photographed by Floyd E. Hayes, B and D by Brad J. Barnwell).

south in a campground. The annual number of nests at all major colony sites of Black-crowned Night Herons combined from 2009–2024 did not change significantly ($F_{1,10} = 0.04$, $P = 0.849$, n = 11 y; Fig. 4). The annual number of nests at all major colony sites combined and water level were not significantly related for any species ($F_{1,7–12} = 0.02–1.17$, $P = 0.300–0.900$; n = 9–14 y).

Discussion

Despite the limitations of our surveys (not all colony sites surveyed annually, surveys occurring at different stages of the breeding season, and more than one survey in a breeding season for some colony sites), our data reveal considerable fluctuations in the presence and number of nests of each species at each colony site. The higher numbers of Double-crested Cormorant and Great Blue Heron nests in 1993 and 1994 (Wolfe and Norman 1998) compared with our surveys during 2006–2024 revealed a significant decrease in the number of nests for each of these species, in contrast with the dramatic increase in breeding populations of the Western Grebe and Clark's Grebe during 1992–2019 (Hayes et al. 2022). The large numbers of nesting Double-crested Cormorants in 1895 (Chamberlin 1895) suggest that its breeding population prior to the 20th Century was even higher. These negative

TABLE 2. Number of years surveyed and active, mean (SD) number of nests, and range for breeding cormorants and herons at different colony sites at Clear Lake and associated wetlands during 1993–2024. Notes and references are a - includes 1999 (Shuford 2014, Shuford et al. 2020b), b - includes 1993–1994 (uncertain which year; Wolfe and Norman 1998), 1998, and 1999 (Shuford 2014, Shuford et al. 2020b), c - includes 1993 and 1994 (Wolfe and Norman 1998), d - includes 1993–1994 (uncertain which year; Wolfe and Norman 1998).

Species and colony site	Years surveyed	Years active	Mean (SD)	Range
Double-crested Cormorant				
Upper Rodman Slough	9	7	22.7 (21.6)	0–53
Reeves Point a	13	8	37.6 (59.4)	0–200
Long Tule Point a	17	6	15.4 (27.4)	0–100
Quercus Point b	12	5	67.5 (89.5)	0–210
Cache Creek a	14	4	3.1 (6.8)	0–22
Clearlake Oaks Wetlands	5	5	26.4 (7.8)	21–40
Great Blue Heron				
Upper Rodman Slough c	11	10	39.0 (37.8)	0–121
Lyons Creek	1	1	15	—
Reeves Point	10	5	6.0 (9.0)	0–27
Long Tule Point	14	14	34.6 (32.4)	1–85
Quercus Point c	8	6	19.9 (36.6)	0–100
Kelsey Creek	18	4	0.6 (1.9)	0–8
Cache Creek c	14	14	36.0 (17.7)	8–65
Clearlake Oaks Wetlands	5	5	9.4 (3.4)	6–14
Great Egret				
Upper Rodman Slough	9	5	3.8 (5.5)	0–15
Lower Rodman Slough	9	4	3.0 (5.2)	0–16
Reeves Point	10	2	1.3 (2.8)	0–7
Quercus Point	6	1	0.2 (0.4)	0–1
Cache Creek	12	11	7.1 (5.8)	0–20
Clearlake Oaks Wetlands	5	1	1.0 (2.2)	0–5
Black-crowned Night-Heron				
Willow Point	12	9	47.4 (37.5)	0–102
Reeves Point	11	1	0.5 (1.5)	0–5
Quercus Point d	7	1	0.4 (1.1)	0–3
Cache Creek	12	2	0.8 (1.8)	0–5
Clearlake Oaks Wetlands	5	1	2.2 (4.9)	0–11

trends contrast with the relatively stable or increasing populations of breeding Double-crested Cormorants and Great Blue Herons in the San Francisco Bay region and elsewhere in California (Kelly et al. 2007; Capitolo et al. 2019; Rauzon et al. 2019; Shuford et al. 2020a,b). If the two long-term trends are genuine rather than artifacts of a small sample size from a population that varies stochastically over time, we do not understand the environmental drivers of these trends.

Because breeding population surveys of cormorants and herons did not begin until more than four decades after the last application of DDD on Clear Lake, the immediate and short-term effects of DDD are unknown. Although DDD concentrations in feathers, tissues, and eggs of grebes at Clear Lake steadily declined in the 1960s and early 1970s (Craig and Rudd 1974), no such studies were conducted on cormorants and herons at the lake until 1993–1994, when much lower and sublethal concentrations of DDD and dichlorodiphenyldichloro-ethylene (DDE, resulting from the breakdown of the related pesticide dichlorodiphenyltrichloroethane or DDT) were found in the tissues of the Double-crested Cormorant and Great Blue Heron (Wolfe and Norman 1998). These studies revealed a gradual multidecadal decline in the organochlorine pesticides in the lake, which clearly did not contribute to the recent declines of Double-crested Cormorants and Great Blue Herons since 1993.
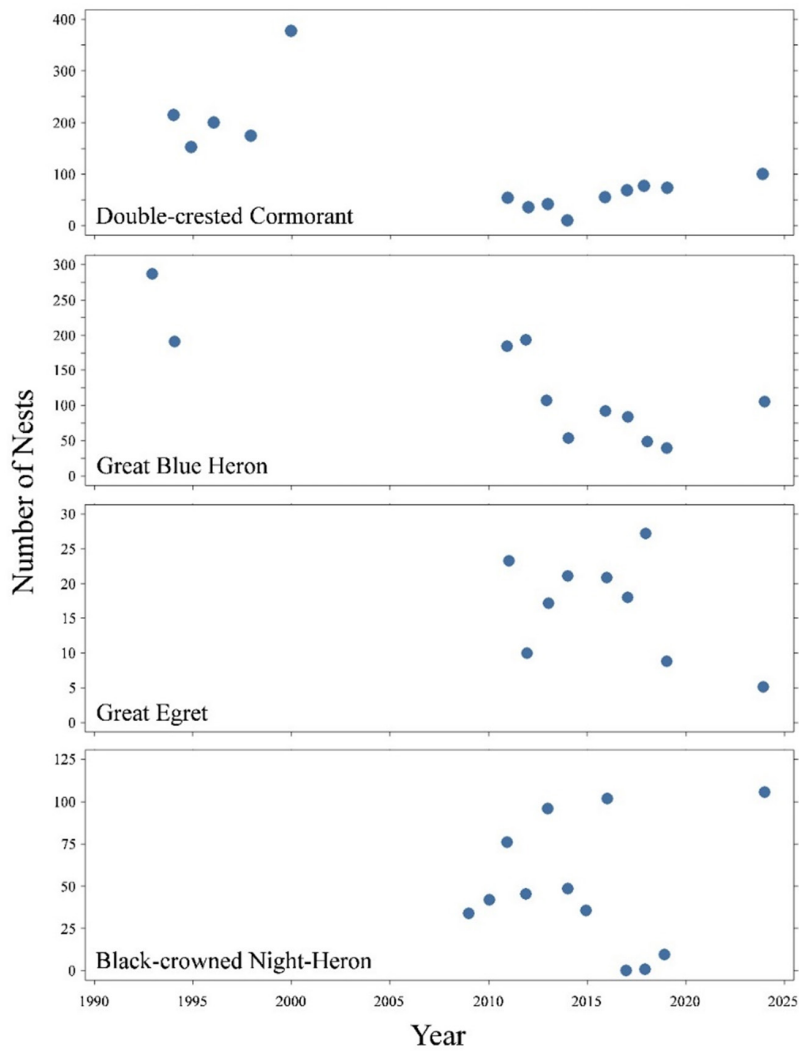
FIGURE 4. Total number of nests per breeding season for Double-crested Cormorant (*Nannopterum auritum*), Great Blue Heron (*Ardea herodias*), Great Egret (*Ardea alba*), and Black-crowned Night-Heron (*Nycticorax nycticorax*), during years when all major colonies in Rodman Slough, Upper Arm, and Cache Creek, Clear Lake, California, were surveyed during 1993–2024.

Mercury (Hg) contamination may also have adversely impacted breeding cormorants and herons. Hg was mined intermittently from the nearby Sulphur Bank Mercury Mine (Fig. 1) during 1872–1957, with increased seepage into the lake after large-scale open pit mining began in 1927 (Suchanek et al. 2000, 2008). Elevated but sublethal Hg concentrations were found in the tissues and feathers of Double-crested Cormorants and Great Blue Herons at Clear Lake in 1993 (Cahill et al. 1997, 1998; Wolfe and Norman 1998), but the growth rates of Great Blue Heron nestlings did not differ from nestlings at other locations uncontaminated with Hg (Wolfe and Norman 1998), suggesting that reproductive effort and success were unaffected by Hg. Hg concentrations in the feathers of another piscivorous bird at Clear Lake, the Western Grebe, declined precipitously from 1967–1969 to 1992, but marked fluctuations occurred in the feathers of the Western Grebe and another piscivorous bird, the Osprey (*Pandion haliaetus*), during 1992–2006,

perhaps due to population fluctuations of their fish prey (Anderson et al. 2008; Eagles-Smith et al. 2008). Hg is presumably declining in cormorants and herons at the lake and is unlikely to have contributed to the decline of breeding Double-crested Cormorants and Great Blue Herons since 1993.

Habitat loss may have reduced the carrying capacity of breeding cormorants and herons. An estimated 85% of the natural wetlands of Clear Lake have been destroyed (Richerson et al. 1994; Suchanek et al. 2003), but we are unaware of any major wetland losses in recent decades. At Clear Lake, cormorants and herons depend on tall, broad-leaved trees for nesting, but no major loss of trees has occurred at any of the 10 colony sites. Thus, habitat loss is unlikely to have contributed to the recent declines since 1993 of breeding Double-crested Cormorants and Great Blue Herons.

Population fluctuations of fish species preyed upon by cormorants and herons potentially affect the

breeding populations. Several dramatic fluctuations in fish populations in Clear Lake have been documented, driven in part by the introduction of exotic species and cold winter temperatures during some years (Eagles-Smith 2008; Thompson et al. 2014). It is possible that the high numbers of nesting Double-crested Cormorant and Great Blue Heron on Clear Lake during 1993–1994 (Wolfe and Norman 1998) were the consequence of a population spike of their fish prey. The Mississippi Silverside (*Menidia audens*) and Threadfin Shad (*Dorosoma petenense*) are small, introduced species that could be a large component of the diet of these birds. Populations of these two abundant fish species, however, were relatively low during 1993–1994 (Eagles-Smith et al. 2008). Unfortunately, no published data are available on their populations since 2004, so we cannot assess the impact of fluctuating fish populations on breeding populations of cormorants and herons.

Cyanobacterial and algal blooms occur frequently during the warm summer months on Clear Lake (Smith et al. 2023) and can affect the spatial distribution of fish populations, especially during episodes of hypoxia. Several studies have documented the avoidance of hypoxia by fish at Clear Lake (Feyrer et al. 2020; Stang 2020; Cortés et al. 2021), which could reduce the availability of fish for piscivorous birds if they move farther from a colony, increasing the energetic expense of foraging. Hypoxia may explain why grebes occasionally abandon their colonies at Clear Lake (Hayes et al. 2022), but the breeding season of cormorants and herons at Clear Lake peaks in April and early May, with most nestlings fledging by the end of May, usually before hypoxic conditions occur.

Fluctuating water levels may affect the distribution of nesting colonies and reproductive effort. Lower water levels exacerbate cyanobacterial and algal blooms, creating more hypoxic conditions as discussed above. Lower water levels may also concentrate fish and other aquatic prey, however, which may be captured more efficiently, potentially increasing the number of nesting pairs. At Clear Lake, colonies of the Western Grebe and Clark's Grebe, which construct floating nests, are more likely to nest in marshes, especially in associated wetlands, when the water level is high, but the number of nests per breeding season was unrelated to water level (Hayes et al. 2022). The breeding season of cormorants and herons at Clear Lake coincides with relatively high-water levels, which decline as the breeding season progresses. Our data demonstrate that the number of cormorant and heron nests was unrelated to water level.

Undetected natural or anthropogenic disturbances may adversely affect the reproductive effort and success of cormorants and herons. Increases in the volatility or amount of rainfall adversely affects reproductive effort of herons in the San Francisco Bay area (Kelly and Condeso 2014) and likely impact their breeding in Clear Lake as well. Several species of birds and mammals prey on the eggs or nestlings of herons and cormorants in the San Francisco Bay area, where the Common Raven (*Corvus corax*) is the dominant predator (Hothem and Hatch 2004; Kelly et al. 2005, 2007; Brussee et al. 2016; Carle et al. 2017). Although we never observed predation on eggs or nestlings of breeding cormorants or herons at Clear Lake, the American Crow (*Corvus brachyrhynchos*) is the dominant diurnal predator and the Northern Raccoon (*Procyon lotor*) is the dominant nocturnal predator of Western Grebe and Clark's Grebe eggs at Clear Lake (Hayes et al. 2022). The raven, crow, and raccoon are human commensals with increasing populations (Marzluff et al. 2001; Kelly et al. 2002; Prange et al. 2003) and may adversely affect reproductive effort and success of cormorants and herons. Frequent disturbances or even shooting by humans could also reduce breeding populations, although we are unaware of any such incidents.

In conclusion, the apparent decline in breeding populations of Double-crested Cormorant and Great Blue Heron since 1993 is difficult to understand. Our data represent a historical baseline for future comparisons. Future monitoring of breeding cormorants and herons, as well as their habitat and prey, is needed to better understand long-term trends and the environmental drivers of change in their breeding populations, especially when new threats emerge, such as the introduction of freshwater mussels (MacIsaac 1996; Nalepa and Schloesser 2014; Karatayev et al. 2015). Such monitoring should occur on an annual basis, preferably more than once per breeding season to determine the dates when nesting peaks. For example, our repeated surveys during some years suggest that the number of nests peaks by mid-April for the Great Blue Heron, by mid-May for the Great Egret, and by late May for the Black-crowned Night-Heron, with interannual variation. Given the challenges of counting nests from the shore or watercraft, videos taken by unmanned aerial systems (often referred to as drones) could be used to supplement counts of nests (Barr et al. 2018; Jones et al. 2020; Prosser et al. 2022).

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BRAD BARNWELL grew up in southern California and graduated from the University of Oregon, Corvallis, with a Parks and Recreation degree. After 5 y in the U.S. Coast Guard and two summers working in Colorado for the U.S. National Park Service, he moved to Lakeport, California, in 1982. He served on the Board of Directors for the Lake County Land Trust and Clear Lake State Park Interpretive Association. As a member of the Redbud Audubon Society, Brad often led hikes and gave presentations, and assisted with a study on the grebes of Clear Lake. Since retiring in 2005, he often observes and photographs birds and wildlife and leads bird walks for the Land Trust, Clear Lake State Park, and Redbud Audubon Society. (Photographed by Brad Barnwell).

DONNA MACKIEWICZ is an avid naturalist with a passion for birding and environmental education. She served as a National Audubon Society board member for 40 y, sponsoring four winning Presidential Environmental Youth Award projects with students in EPA Region 6, and currently serves as president of the Redbud Audubon Society. Currently a member of the Blue Ribbon Committee for the Rehabilitation of Clear Lake, her work on the Socio-economic Committee keeps her focused on the long term brighter future for Clear Lake. (Photographed by Roberta Lyons).

APPENDIX TABLE. Survey dates and number of nests at each colony site where each cormorant and heron species was detected breeding during 2006–2024, and in sites (indicated with asterisk) where breeding was detected during 1993–2000 by others but not during 2006–2024.

Double-crested Cormorant (*Nannopterum auritum*)

Upper Rodman Slough
2011 – 03 March, 20; 10 April, 53; 22 May, 35
2012 – 08 April, 32; 22 May, 23
2013 – 14 April, 38
2014 – 20 April, 6
2016 – 24 April, 51
2017 – 30 April, 21
2018 – 29 April, 3
2019 – 28 April, 0
2024 – 21 March, 0; 11 April, 0

Reeves Point
2006 – 22 April, 100
2008 – 26 April, 2
2009 – 25 April, 3
2010 – 28 March, 0; 24 April, 0
2011 – 10 April, 0
2012 – 13 May, 0
2014 – 20 April, 0; 20 May, 0
2018 – 29 April, 50
2019 – 28 April, 41
2024 – 21 March, 30

Long Tule Point
2006 – 22 April, 0
2008 – 26 April, 0
2009 – 12, 25 April, 0
2010 – 28 March, 0; 24 April, 0
2011 – 20 March, 0; 10 April, 0; 22 May, 0
2012 – 08 April, 0; 13 May, 0
2013 – 14 April, 0
2014 – 20 April, 0
2015 – 02 May, 0
2016 – 24 April, 0
2017 – 30 April, 25
2018 – 29 April, 20
2019 – 28 April, 29
2024 – 21 March, 31

Quercus Point*
2006 – 22 April, 0
2008 – 26 April, 0
2009 – 25 April, 0
2010 – 24 April, 0
2011 – 10 April, 0
2024 – 21 March, 0

Cache Creek
2009 – 05, 26 April, 0

2010 – 28 March, 0; 02 May, 0
2011 – 20 March, 0; 10 April, 0; 22 May, 0
2012 – 08 April, 0; 05 May, 0
2013 – 14 April, 0
2014 – 20 April, 0
2015 – 02 May, 0
2016 – 24 April, 0
2017 – 30 April, 22
2018 – 12 April, 0; 06 May, 2
2019 – 18 April, 0; 27 May, 4
2024 – 11 April, 0

Clearlake Oaks Wetlands
2020 – 26 March, 0; 30 April, 22
2021 – 02 March, 24; 03 March, present
2022 – 03 March, 0; 15 March, 25
2023 – 12 January, 12; 14 January 20; 01 February, 21
2024 – 01 January, 15; 01 February, 28; 01 March, 30; 01 April, 40; 30 April, present

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Great Blue Heron (*Ardea herodias*)

Upper Rodman Slough
2011 – 03 March, 27; 10 April, 46; 22 May, 33
2012 – 08 April, 50; 22 May, 10
2013 – 14 April, 23
2014 – 20 April, 16
2016 – 24 April, 42
2017 – 30 April, 19
2018 – 29 April, 12
2019 – 28 April, 7
2024 – 21 March, 0; 11 April, 0

Lyons Creek
2024 – 14 April, 15

Reeves Point
2006 – 22 April, 10
2008 – 26 April, 1
2009 – 25 April, 0
2010 – 28 March, 0; 24 April, 0
2011 – 10 April, 0
2012 – 13 May, 0
2014 – 20 April, 0; 20 May, 0
2018 – 29 April, 7
2019 – 28 April, 9
2024 – 21 March, 13

Long Tule Point
2006 – 22 April, 20
2008 – 26 April, 12
2009 – 12 April, 57
2010 – 28 March, 85; 24 April, present
2011 – 20 March, 71; 10 April, 84; 22 May, 24
2012 – 08 April, 84; 13 May, present
2013 – 14 April, 61
2014 – 20 April, 25
2015 – 02 May, 27
2016 – 24 April, 8
2017 – 30 April, 10
2018 – 29 April, 1
2019 – 28 April, 3
2024 – 21 March, 8

Quercus Point
2006 – 22 April, 0
2008 – 26 April, 2
2009 – 25 April, 0
2010 – 24 April, 5
2011 – 10 April, 2
2024 – 21 March, 0

Kelsey Creek
2021 – 6, 11 April, 1
2022 – 12, 14, 20 March, 1
2023 – 9, 20 February, 1; 24, 27 March, 1; 24 April, 1;
13, 16, 18, 25, 27 May, 1; 1 June, 1
2024 – 6, 15 January, 1; 9 February, 3; 12 February,
5; 25 February, 7; 25, 31 March, 7; 11, 15 April, 7;
25 April, 8

Cache Creek
2009 – 05 April, 19; 26 April, present
2010 – 28 March, 52; 02 May, present
2011 – 20 March, 53; 10 April, 52; 22 May, 13
2012 – 08 April, 57; 05 May, present
2013 – 14 April, 22
2014 – 20 April, 11
2015 – 02 May, 25
2016 – 24 April, 44
2017 – 30 April, 41
2018 – 12 April, 19; 06 May, 27
2019 – 18 April, 8; 27 May, 6
2024 – 11 April, 35

Clearlake Oaks Wetlands
2020 – 26 March, 6; 30 April, present
2021 – 02 March, present; 03 March, 7
2022 – 03 March, 8; 15 March, present
2023 – 12, 14 January, 12; 1 February, 14
2024 – 01 January, 0; 01 February, 0; 01 March, 12;
01 April, present; 30 April, 12

Great Egret (*Ardea alba*)

Upper Rodman Slough
2011 – 03 March, 0; 10 April, 7; 22 May, 9
2012 – 08 April, 0; 22 May, 1
2013 – 14 April, 0
2014 – 20 April, 0
2016 – 24 April, 15
2017 – 30 April, 8
2018 – 29 April, 1
2019 – 28 April, 0
2024 – 21 March, 0; 11 April, 0

Lower Rodman Slough
2011 – 03 March, 0; 10 April, 3; 22 May, 3
2012 – 08 April, 0; 22 May, 5
2013 – 14 April, 3
2014 – 20 April, 16
2016 – 24 April, 0
2017 – 30 April, 0
2018 – 29 April, 0
2019 – 28 April, 0
2024 – 21 March, 0; 11 April, 0

Reeves Point
2006 – 22 April, 0
2008 – 26 April, 0
2009 – 25 April, 0
2010 – 28 March, 0; 24 April, 0
2011 – 10 April, 0
2012 – 13 May, 0
2014 – 20 April, 0; 20 May, 0
2018 – 29 April, 6
2019 – 28 April, 7
2024 – 21 March, 0

Quercus Point
2006 – 22 April, 0
2008 – 26 April, 0
2009 – 25 April, 1
2010 – 24 April, 0
2011 – 10 April, 0
2024 – 21 March, 0

Cache Creek
2009 – 05, 26 April, 1
2010 – 28 March, 2; 02 May, 7
2011 – 20 March, 0; 10 April, 9; 22 May, 11
2012 – 08 April, 2; 05 May, 4
2013 – 14 April, 14
2014 – 20 April, 5
2015 – 02 May, 4
2016 – 24 April, 6
2017 – 30 April, 10

2018 – 12 April, 20; 06 May, present
2019 – 18 April, 1; 27 May, 2
2024 – 11 April, 35

Clearlake Oaks Wetlands
2020 – 26 March, 0; 30 April, 0
2021 – 03 March, 0
2022 – 03 March, 0; 15 March, 0
2023 – 12, 14 January, 0; 1 February, 0
2024 – 01 January, 0; 01 February, 0; 01 March, 5; 01, 30 April, present

Black-crowned Night-Heron (*Nycticorax nycticorax*)

Willow Point
2009 – 17 May, 34
2010 – 6 June, 41
2011 – 20 March, 0; 10 April, 0; 22 May, 41
2012 – 10 June, 41
2013 – 14 April, 0; 21 May, present; 14 June, 96
2014 – 20 April, 0; 20 May, 49
2015 – 2 May, 36
2016 – 24 April, 102
2017 – 30 April, 0
2018 – 29 April, 0
2019 – 28 April, 0
2024 – 11 April, 94; 14 April, 95

Reeves Point
2006 – 22 April, 0
2008 – 26 April, 0
2009 – 25 April, 0
2010 – 28 March, 0; 24 April, 0
2011 – 10 April, 0

2012 – 13 May, 0
2014 – 20 April, 0; 20 May, 0
2018 – 29 April, 0
2019 – 28 April, 5
2024 – 21 March, 0

Quercus Point*
2006 – 22 April, 0
2008 – 26 April, 0
2009 – 25 April, 0
2010 – 24 April, 0
2011 – 10 April, 0
2024 – 21 March, 0

Cache Creek
2009 – 05, 26 April, 0
2010 – 28 March, 0; 02 May, 0
2011 – 20 March, 0; 10 April, 0; 22 May, 0
2012 – 08 April, 0; 05 May, 4
2013 – 14 April, 0
2014 – 20 April, 0
2015 – 02 May, 0
2016 – 24 April, 0
2017 – 30 April, 0
2018 – 12 April, 0; 06 May, 0
2019 – 18 April, 0; 27 May, 5
2024 – 11 April, 0

Clearlake Oaks Wetlands
2020 – 26 March, 0; 30 April, 0
2021 – 03 March, 0
2022 – 03 March, 0; 15 March, 0
2023 – 12, 14 January, 0; 1 February, 0
2024 – 01 January, 0; 01 February, 0; 01 March, 0; 01, 30 April, 8

Aspidoscelis velox (Plateau Striped Whiptail) in Jefferson County, Oregon, USA: Persistence of an Introduced Triploid Parthenogenetic Species of Lizard

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Abstract.—We have studied the triploid parthenogenetic Plateau Striped Whiptail (*Aspidoscelis velox*) in many parts of its natural distribution area in Arizona, Colorado, New Mexico, and Utah, USA. In 1983 it was reported to have become established in Cove Palisades State Park in Jefferson County, Oregon, USA. That introduction is believed to have occurred before 1970. The species also has been identified at newly discovered sites of introduction east of the Continental Divide in Colorado. Most parthenogenetic species of whiptail lizards in the USA have been discovered in areas beyond their natural distributions. Establishment of a parthenogen requires survival of successive generations of young-of-the-year, minimally starting with a single introduced lizard or egg, to found a new array. In July 2023 we undertook a survey to assess the status of *A. velox* in Cove Palisades State Park, Jefferson County, Oregon, 55–60 years after its introduction. We observed 10 whiptail lizards over 2 d. We mapped these observations as well as verifiable records from iNaturalist.org. Recent observations south of Cove Palisades State Park suggest this introduced array not only has persisted, but is continuing to expand its local distribution.

Key Words.—State Park; Oregon; introduced array; triploid; parthenogenetic; whiptail lizards

Introduction

The triploid parthenogenetic Plateau Striped Whiptail (*Aspidoscelis velox*) has a natural distribution area on and near the Colorado Plateau in Arizona, Colorado, New Mexico, and Utah, USA. Apparently, it was first reported in the herpetofauna of Oregon by Nussbaum et al. (1983), who suggested the species had been introduced at least six years previously. Subsequently, Storm et al. (1995) stated that the lizard was introduced to central Oregon sometime prior to 1970. They reported that it inhabited rocky juniper-grown areas on the west side of Lake Billy Chinook in Cove Palisades State Park. Halvorson (2004) suggested the species expanded its range in the park after an individual escaped from a recreational vehicle. The gonochoristic Great Basin Whiptail (*Aspidoscelis tigris tigris*), which has a natural distribution that includes eastern Oregon, is the only species of its genus in the state in addition to *A. velox*. We provide an update on the status of the array (i.e., we reserve the term population for gonochoristic species) of triploid parthenogenetic *A. velox* that we think has persisted in Oregon for as long as 55–60 y after its deliberate or accidental introduction in the state. As of this writing, 16 observations (with images) of *A. velox* from Oregon also have been reported on iNaturalist.org between 1999 and August 2024.

Materials and Methods

We studied images of whiptail lizards from Oregon posted on iNaturalist.org through August 2024 and agree that images identified as either the Western Whiptail (*Aspidoscelis tigris*) or Great Basin Whiptail (*A. tigris tigris*) were correctly identified as either of these two taxa based on distinctive dorsal color patterns. Similarly, all posted images identified as the Plateau Striped Whiptail (*Aspidoscelis velox*) on iNaturalist.org were correctly identified based on distinctive dorsal color patterns. The dorsal color pattern of *A. velox* includes six distinct light dorsal stripes (when present, a seventh stripe is less distinct and located midorsally) against a dark background. In contrast, *A. tigris tigris* has variation in dorsal pattern that often includes light stripes (often obscured) and brownish or gray spots, bars, or other markings on a dark background (Stebbins 2003). All 16 of the postings of Plateau Striped Whiptail were unquestionably from Jefferson County, where the species was previously known to have been introduced in the state (Nussbaum et al. 1983; Stebbins 1985; Storm et al. 1995). We have included additional details in Fig. 1 and the Appendix. Two records on iNaturalist.org had obscured coordinates so, although the county was listed, the location could not be pinpointed. Of the remaining 14 postings, only three

images of *A. velox* from Jefferson County seemed to be from outside of Cove Palisades State Park; namely two from northwest of the city of Terrebonne and one from Crooked Creek River National Grassland. The latter record had a large estimated coordinate error (11.73 km) so that the precise location of the record relative to Cove Palisades State Park is uncertain.

Our field survey to determine the current status of the descendants of an introduced array of *A. velox* in Cove Palisades State Park was conducted by CMK and KLU at the request and support of JMW. Other authors were recruited based on their abilities to contribute in various components of the study. The state park, located in north-central Oregon, encompasses the Deschutes and Crooked River Canyons. A prominent feature of the park is its proximity to Lake Billy Chinook (Fig. 1). The areas surveyed in the park from about 0900–1400 during 17–18 July 2023 were in the vicinity of the Deschutes Campground and the Tam-a-láu Trail (Fig. 2).

The survey method used was similar to time constrained Visual Encounter Surveys (VES) (Campbell and Christman 1982; Corn and Bury 1990). Surveys were one hour each, and sites were surveyed twice per day at: (1) B loop campground; (2) Tam-a-láu trail; and (3) area south of the Tam-a-láu trail. At each site, observers walked 5–10 m apart. At the B loop campground, the road through the campground was used as a transect, with observers walking on opposite sides of the road. On the Tam-a-láu trail, each observer walked approximately 1–5 m on each side of the trail. On the area south of the trail, observers walked in straight lines as the area was somewhat rectangular and flat. When a lizard was encountered, a single observation was conducted by two observers from approximately 5 m away, allowing for unbroken visual contact.

Areas that were found to be inhabited by whiptail lizards were characterized by a relatively flat landscape with openings among an abundance of shrubs, grasses, and woody debris. Sections of the Tam-a-láu Trail were found to be especially productive for *A. velox*, though steeply inclined sections of the trail were not searched (Fig. 2). We recorded the following data when an individual *A. velox* was spotted: (1) air temperature from NOAA National Weather Service (https://www.weather.gov/) for Jefferson County, Oregon; (2) whether the sky was clear or cloudy; (3) time of day; (4) estimated age class of the lizard based on snout-vent length (SVL) with lizards > 70 mm SVL classified as adults and smaller individuals classified as juveniles (Persons and Wright 2009); (5) dominant plant species (identified subsequently); and (6) general habitat structure. Habitat structure parameters included (1) anthropogenic structures (e.g., bathrooms, road, picnic table), (2) shrubs, (3) trees, (4) grasses, (5) dead wood fragments on the ground, and (6) topography (e.g., flat versus steep slope by visual assessment). We also noted predation attempts. The general locations of the observations were also assessed
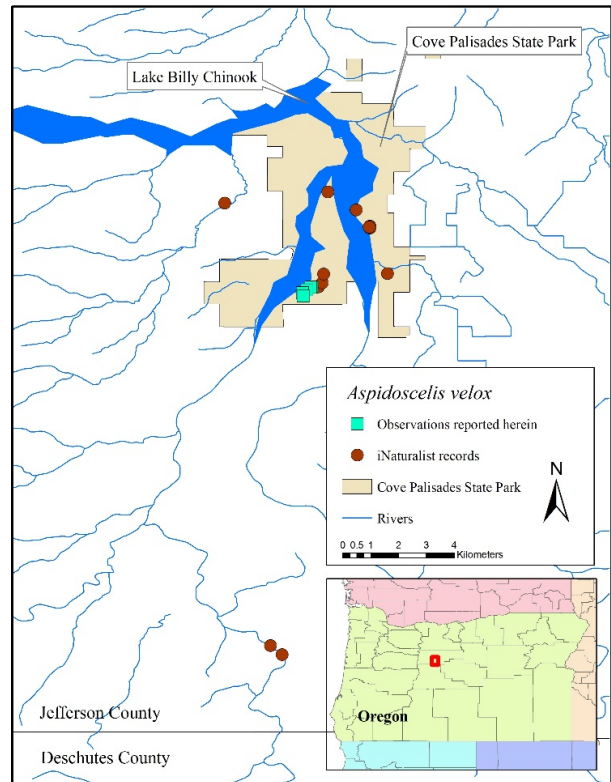

Figure 1. Distribution of 14 of 16 triploid parthenogenetic Plateau Striped Whiptail (*Aspidoscelis velox*) reports on iNaturalist.org in Jefferson County, Oregon, and the location of observations reported in this article. Some iNaturalist.org records occurred in close enough proximity for the records to overlap, and two reports had obscured coordinates and are not mapped. The westernmost iNaturalist observation, plotted outside Cove Palisades State Park, had an 11.73 km coordinate error and was thus possibly also from within the park.

relative to the location of the Deschutes Campground. We captured several lizards using a lasso attached to an extendable pole, after which we photographed each lizard with a digital camera. We removed a small sample of tissue from the end of the tail of captured lizards and immediately immersed the sample in 95% ethanol in a separate vial for each lizard for a separate analysis. We released all captured individuals of *A. velox* within 10 min of capture near the point of their initial observation.

RESULTS

Whiptail lizards observed and captured in the complex landscape in Cove Palisades State Park were readily identifiable as *A. velox* based on distinctive color patterns (Figs. 3–4). In addition, all images posted prior to September 2024 on iNaturalist.org from Oregon under the names *A. velox*, *A. tigris*, and *A. t. tigris* were correctly identified. The six captured individuals examined and photographed had ventral colorations of pale blue (Fig. 3A) to darker blue with a slight grayish tinge in the central parts of the scales in the eight longitudinal rows (Fig. 3B). We attributed differences
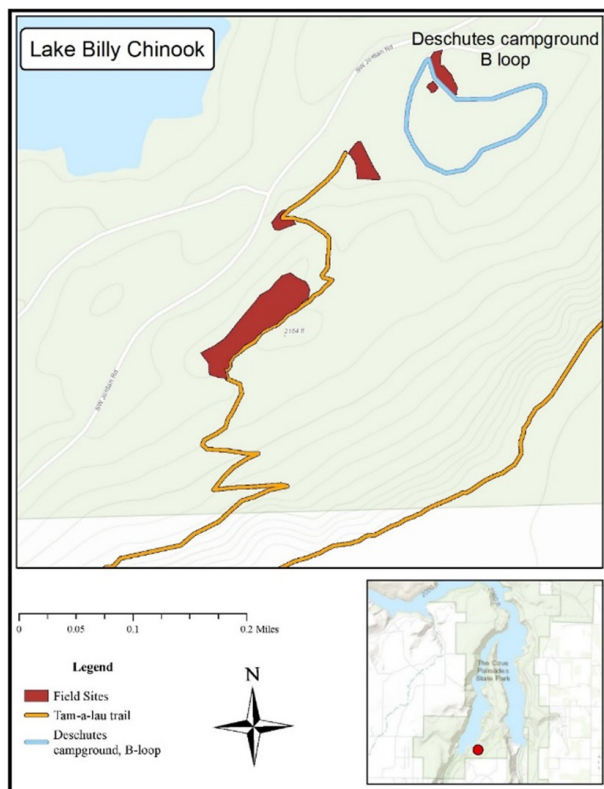
Figure 2. Map of the Deschutes campground, relative to Lake Billy Chinook, and the Tam-a-láu Trail (orange), Cove Palisades State Park, Jefferson County, Oregon. The red polygons represent the flat areas of habitat surveyed where adults of triploid parthenogenetic Plateau Striped Whiptail (*Aspidoscelis velox*) were observed (some were lassoed). The Cove Palisades State Park and the Deschutes campground (red dot) can be seen in the map in the bottom right corner.

in the hues of the ventral coloration of adult lizards to ontogenetic variation, with the blue intensifying with age. All individuals both observed and captured had six distinct primary dorsal stripes arranged on the trunk in ventral to dorsal bilateral pairs of whitish laterals, cream-white dorsolaterals, and slightly darker hued cream-white paravertebrals (Figs. 3–4). The only indication of development of a seventh stripe (= secondary middorsal line) between the paravertebrals was a barely visible vestigial vertebral fragment from the occipital scales on the head posteriorly to near the level of the forelimbs (Fig. 3C) and a ghost-like vertebral (= middorsal) line along the length of the trunk (Fig. 4C). Though interrupted by the hindlimbs, bilateral stripes equivalent to the laterals of the trunk were present on the tail, as were extensions of the paravertebrals extending from the trunk (Fig. 4D). The longitudinal fields between the stripes on the trunk and basal third of the tail were black in juveniles and young adults (Fig. 4D). Older adults had slightly less vividly contrasting stripes and fields though there was no evidence of stripe disruption into spots (Figs. 4B–C). None of the lizards had distinct spots in the longitudinal dark-hued fields between the stripes although the central

regions of the fields in older lizards were slightly lighter in color than the outer regions. The largest adults did possess fields between the stripes that trended toward a dark brown coloration which contributed to reduced contrast between stripes and fields. We have examined data for large numbers of specimens of *A. velox* from Arizona, Colorado, New Mexico, and Utah, and have found significant geographic variation in the development of a middorsal line ranging from a dash of a few mm extending from the occipital scales (e.g., from several sites in Arizona) to presence of a distinct seventh stripe (e.g., specimens of all sizes from Kane County, Utah, and Cibola County, New Mexico; Walker 1986; Cordes and Walker 2013; Sullivan et al. 2018; Cole et al. 2019; Livo 2020). Taylor (1965) also reported ontogenetic variation in this character in specimens from Colorado.

The landscape in Cove Palisades State Park was primarily populated by conifers, juniper trees (*Juniperus* sp.), Great Basin Sagebrush (*Artemisia tridentata*), Gray Rabbitbrush (*Ericameria nauseosa*), Green Rabbitbrush (*Chrysothamnus viscidiflorus*), and various grasses (Fig. 5). On 17–18 July 2023, weather conditions were clear and sunny. The earliest time of observation of an adult whiptail lizard was 0853 at an air temperature of 17.2° C (63° F) in a shrub dominated area near the Tam-a-láu trailhead. One individual *A. velox*, which was captured by lasso, was shedding, indicating significant prior growth. The other nine individuals (one juvenile, two young adults, and six older adults) were invariably initially observed near environmental structures such as shrubs, juniper or other trees, and piles of deadwood, rather than in more open areas. Other details of the encounters were that air temperatures were between 23.3° and 26.1° C (74° and 79° F), and lizards were found between 1115 and 1251 with cloudless skies. Behaviorally, individuals of *A. velox* were not especially wary and typically could be followed, photographed, or lassoed with relative ease. Individual *A. velox* did not hesitate to enter debris-strewn microhabitat for foraging, concealment, and escape (i.e., areas without exposed substrate; Fig. 4).

Discussion

We have known that *A. velox* is a triploid species since the report of Pennock (1965), and that it reproduces asexually by parthenogenesis based on analyses and experiments conducted by Maslin (1962, 1964, 1966). The lizard evolved via a two-stage hybridization process many generations apart as reported by Dessauer and Cole (1989), Moritz et al. (1989), and Barley et al. (2021). Cole et al. (2019) found that *A. innotatus* described by Burger (1950) from Kane County, Utah, was triploid, and not the diploid, parthenogenetic maternal progenitor in the second stage of hybridization leading to *A. velox* as suspected by Wright (1993). Consequently, the name *A. innotatus* was returned to the synonymy of *A. velox* by Cole et al. (2019).
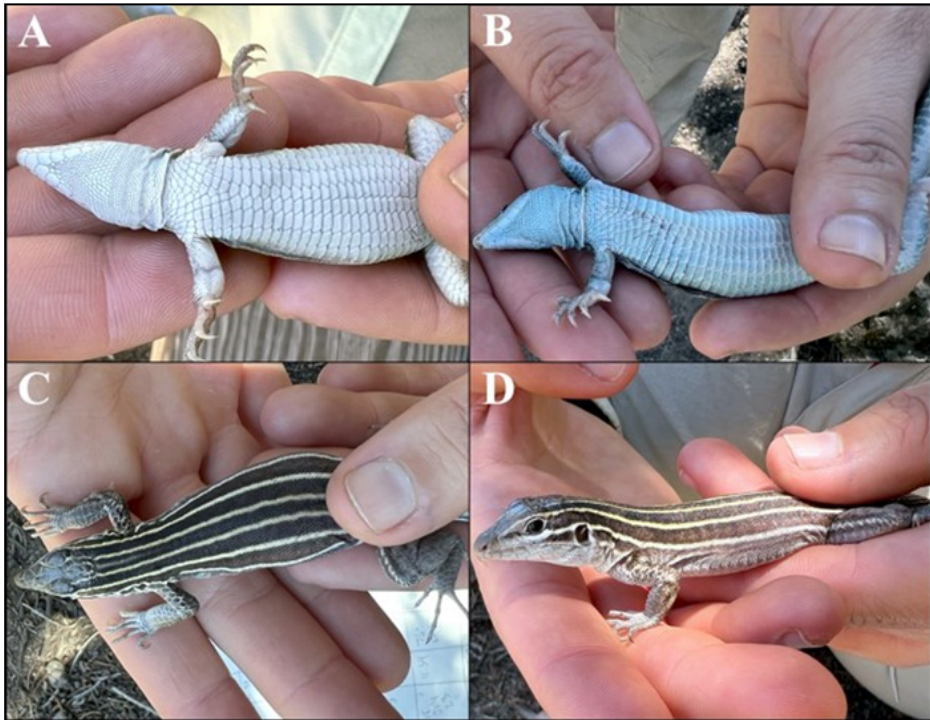
Figure 3. Captured and released adult females of triploid parthenogenetic Plateau Striped Whiptail (*Aspidoscelis velox*) in Cove Palisades State Park, Jefferson County, Oregon, photographed 17–18 July 2023: (A) pale blue-white coloration from ventral perspective of head and trunk; (B) darker blue to blue-gray coloration from ventral perspective of head and trunk; (C) dorsal pattern of six distinctive primary stripes, black intervening fields, and faint vertebral (= middorsal) secondary line from occipital scales to near level of forelimbs; and (D) color pattern from left lateral perspective showing ghost-like markings in the two lowest longitudinal dark fields. (Photographed by Carina M. Kusaka).

Figure 4. Adult females of triploid parthenogenetic Plateau Striped Whiptail (*Aspidoscelis velox*) photographed in situ in Cove Palisades State Park, Jefferson County, Oregon, 17–18 July 2023: (A) part of dorsal pattern on right side of body of an alert lizard in substantial ground cover; (B) dorsal pattern of an older adult lizard showing reduced contrast between longitudinal stripes and fields and gray-blue coloration of distal part of tail; (C) older adult showing semblance of organization of a vertebral (= middorsal) stripe; and (D) part of dorsal pattern from left side of body of young adult with strongly contrasting longitudinal stripes and fields on trunk and tail. (Photographed by Carina M. Kusaka).
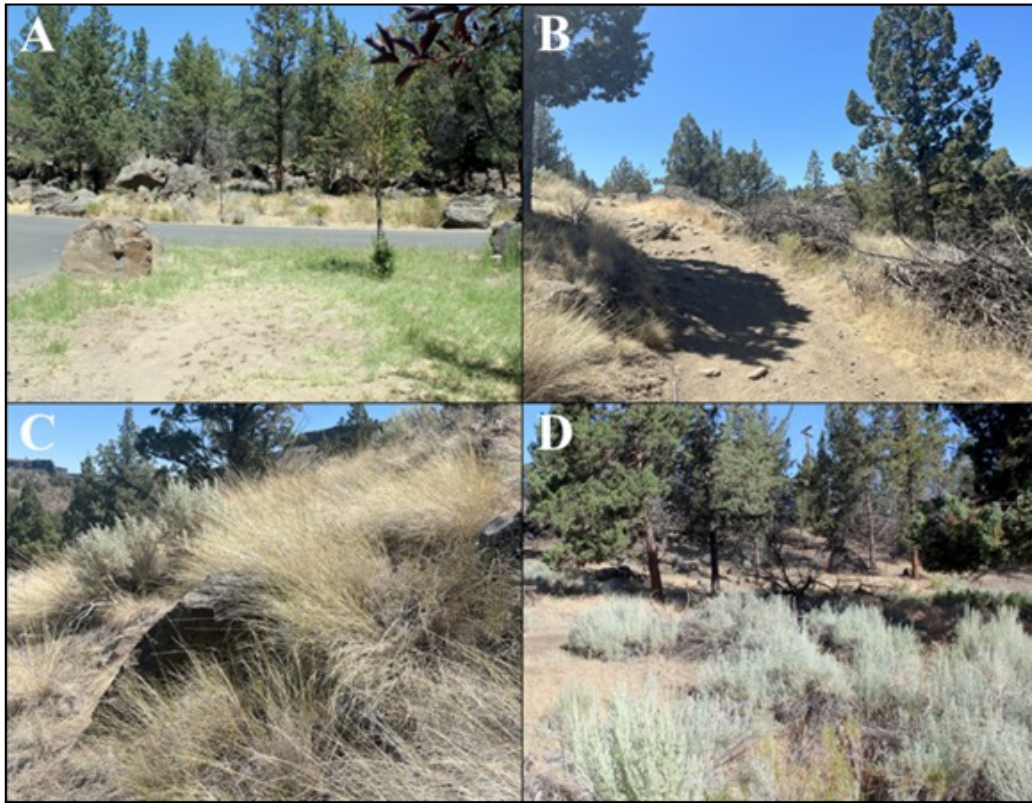
FIGURE 5. Complex landscapes inhabited by triploid parthenogenetic Plateau Striped Whiptail (*Aspidoscelis velox*) in Cove Palisades State Park, Jefferson County, Oregon, photographed 17–18 July 2023. (A) Habitat within Deschutes campground. (B) Loop B approximately 0.48 km into the Tam-a-láu trail where an adult whiptail lizard was found in pile of dead wood on the right side of the trail. (C) Habitat adjacent to the Tam-a-láu trail characterized by various species of grasses among junipers (*Juniperus* sp.) and Great Basin Sagebrush (*Artemisia tridentata*). (D) habitat located south of the Tam-a-láu trailhead with complex vegetational association of *Juniperus* sp., *Artemisia tridentata*, Grey Rabbitbrush (*Ericameria nauseosa*), and Green Rabbitbrush (*Chrysothamnus viscidiflorus*). (Photographed by Carina M. Kusaka).

The identification of the diploid parthenogenetic maternal progenitor of *A. velox* has been an elusive quest. It is thought by some herpetologists to be extinct, although the progenitor may be the species recently described as *Aspidoscelis preopatae* (no common name) from a restricted area in Sonora, México (Barley et al. 2021). The presence of *A. velox* in Cove Palisades State Park is approximately 1,000 km northwest from its natural distribution areas in Arizona, Colorado, New Mexico, and Utah (Nussbaum et al. 1983; Stebbins 1985, 2003; Storm et al. 1995). Oregon, however, does not contain the only extralimital introduced arrays of *A. velox*. Johnson et al. (2020) reported that one or more introductions likely accounts for the presence of the species in the Arkansas River drainage in Chaffee and Fremont counties, Colorado, the only known areas of occurrence for the species in the state east of the Continental Divide.

Aspidoscelis velox is not the only parthenogenetic whiptail lizard species that has been introduced outside its natural distribution. Three other triploid species, the Colorado Checkered Whiptail (*A. neotesselatus*), Chihuahuan Spotted Whiptail (*A. exsanguis*), and Sonoran Spotted Whiptail (*A. sonorae*), are known to have been introduced to Washington State (Weaver et al. 2011), Colorado (Livo and Wilcox 2021), and California (Fisher et al. 2022), respectively. The diploid parthenogenetic New Mexico Whiptail (*A. neomexicanus*) is also known to have become established in Arizona (Persons and Wright 1999; Persons et al. 2021) and Utah (Oliver and Wright 2007; Vicente Fernandez, pers. comm.). Some biologists suggest that *A. neomexicanus* has also been introduced to several areas in New Mexico, including the vicinity of Conchas Lake in San Miguel County (Manning et al. 2005; but see Leuck et al. 1981). Moreover, introduced arrays of the diploid parthenogenetic Gray Checkered Whiptail (*A. dixoni*; Chad Montgomery, pers. comm.), Laredo Striped Whiptail (*A. laredoensis*; Matthew Fujita, pers. comm.), and Common Checkered Whiptail (*A. tesselatus*; Gregory Pauly, pers. comm.) have been found in various parts of the USA west of the Mississippi River.

We note that few groups of vertebrates better illustrate the distinction between ecology and biogeography than species of parthenogenetic whiptail lizards (i.e., constrained to an area by local and/or intervening ecological conditions though able to thrive when introduced to new areas). The array of *A. velox* in Oregon and those of maternal progenitor *A. preopatae* in Sonora

are separated by > 2,000 km of habitats that include woodlands and deserts, which may be the ultimate incongruent relationship between a triploid parthenogen and its diploid maternal progenitor. The geographical closest triploid parthenogenetic species derived from *A. preopatae* as maternal progenitor is the Opata Whiptail (*Aspidoscelis opatae*; Wright 1967), which is also restricted to Sonora.

We have examined samples of *A. velox* from each of the four states within its natural distribution and are reluctant to offer more than a tentative hypothesis on the origin of the pioneers that founded the Oregon array pending examination of genetic samples. The nearest arrays with similar six striped dorsal color patterns, however, are found in Arizona. We note that the nearest arrays of the species, which are in Utah, have seven distinct stripes. Although we will probably never know the circumstances that resulted in the introduction of *A. velox* to Cove Palisades State Park, likely only one or a very small number of individuals initiated this array, which now has endured 55–60 y. Recent observations from more than 12 km south of Cove Palisades State Park suggest that these lizards continue to expand their distribution in the area and land managers should consider surveying these and nearby areas to determine the full current extent of their range in Oregon.

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APPENDIX

We examined several online sources for records of *Aspidoscelis velox*: iNaturalist, Herpetological Education and Research Project (H.E.R.P), Herpmapper (HM), VertNet, and ARCTOS.

No Oregon records were available from VertNet or ARCTOS.

HM: record 128623 from Jefferson County, Oregon cross-listed below as iNaturalist 38392343 and H.E.R.P. 216441.

H.E.R.P.: Four records, all from Jefferson County, Oregon: 198271, 198270, 198269, and 216441 (duplicated as iNaturalist record 38392343 and HM 128623).

In iNaturalist there are 16 observation records for *Aspidoscelis velox* in Jefferson County, Oregon, as of August 2024 (estimated coordinate errors follow record numbers in parentheses): 186498355 (1.36 km), 182533701 (coordinate error not recorded), 162526113 (coordinates obscured), 150897815 (15 m), 150834526 (158 m), 128481093 (32 m), 128102636 (coordinates obscured), 103066603 (50 m), 70019066 (22 m), 51863304 (22 m), 38392343 (15 m), 33111300 (11 m), 32817591 (7 m), 27673680 (11.73 km), 7543674 (coordinate error not recorded), 236619173 (25 m).

Tables in appendices of reports by Schuller and Halvorson (2008a, b) on the Horse Ridge Research Natural Area (Deschutes County) and the Powell Butte Research Natural Area (Crook County), Oregon, list *Aspidoscelis velox*. The presence of species in these tables, however, does not indicate that they are documented for the areas, but as listed are only known or expected to occur.

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Schuller, R., and R. Halvorson. 2008b. Powell Butte Research Natural Area: guidebook supplement 38. General Technical Report PNW-GTR-773, U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. 20 p.

Carina M. Kusaka is a Graduate Research Assistant from Oregon State University in Corvallis, Oregon, USA. Her current research focuses on spatial analyses in the breeding habitat of Tufted Puffins (*Fratercula cirrhata*). She received a B.S. degree in Fish, Wildlife, and Conservation Biology from Colorado State University, Fort Collins, Colorado, where she worked with Erell Insitute to study the behavioral and movement ecology of reptiles. Carina is primarily interested in understanding the effects of climate change on species of greatest conservation need, and using art to help foster excitement and education in communities for their local ecosystems. (Photographed by Donna Grippa).

Kaera L. Utsumi is a Behavioral Ecologist specifically interested in reptile behavior. She is currently a Research Assistant at Erell Institute, Lawrence, Kansas, and has worked with the organization on multiple research projects in several countries engaging women and underrepresented groups in science. Some of the lizards she has studied include the Atacamen Pacific Iguana (*Microlophus atacamensis*), the Colorado Checkered Whiptail (*Aspidoscelis neotesselatus*), and the Namib Web-footed Gecko (*Pachydactylus rangei*). When Kaera is not working on writing papers for publication, she enjoys reading, crocheting, rock climbing, and playing basketball. (Photographed by Maria Eifler).

James E. Cordes is a Professor of Biology at Louisiana State University Eunice (LSUE), Louisiana. He received B.S and M.S. degrees from Texas State University, San Marcos, and Ph.D. from the University of Arkansas, Fayetteville. He has been the recipient of 16 annual Endowed Professorships funded by Opelousas General Hospital and awarded by LSU, Eunice, to study the genetic relationships of parthenogenetic teiid lizards through skin-graft experiments. Since 1984, he has undertaken > 100 field expeditions to México, Arizona, Colorado, New Mexico, Oklahoma, Texas, and Utah, USA, to collect live parthenogenetic whiptail lizards for laboratory experiments and preserved voucher specimens of numerous species for ecological and systematic studies. He is author of > 90 publications on lizards in the genus *Aspidoscelis*. (Photographed by Travis Webb).

Anthony J. Barley is an Assistant Professor in the School of Mathematical and Natural Sciences at Arizona State University (ASU), Glendale. He grew up in California, earned a B.S. degree from California State University, Sacramento, and a Ph.D. from the University of Kansas, Lawrence, where he studied the systematics and evolution of Philippine lizards. Before starting his faculty position at ASU in 2022, he was a postdoctoral researcher at the University of Hawai'i, Honolulu, and the University of California, Davis, where he began research on the evolution of North American whiptail lizards, which has involved fieldwork and collaboration with scientists from across the southwestern U.S., Mexico, and Guatemala. (Photographed by Armand Saavedra).

Robert C. Thomson is a Professor in the School of Life Sciences at the University of Hawai'i, Honolulu. He earned a B.S. at the Rochester Institute of Technology, Rochester, New York, followed by a Ph.D. at the University of California, Davis, that focused on the evolutionary history and conservation biology of amphibians and reptiles. Robert works with a network of students, scientists, and conservation managers to understand the generation and maintenance of biodiversity, and to conserve natural populations of amphibians and reptiles. His recent work has focused on the evolution of whiptail lizards, the diversification of turtles, and statistical methods for inferring evolutionary history. (Photographed by Anthony Barley).

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, 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, phenology, and natural history of amphibian and reptile species in Colorado, especially that of various introduced species including the Pond Slider (*Trachemys scripta*), the Colorado Checkered Whiptail (*Aspidoscelis neotesselatus*), *A. velox*, and the Chihuahuan Spotted Whiptail (*A. exsanguis*). (Photographed by Steve Wilcox).

James M. Walker is Professor of Biological Sciences, University of Arkansas, Fayetteville. Since earning B.S. and M.S. degrees from Louisiana Tech University, Ruston, and a Ph.D. from the University of Colorado, Boulder. James has engaged in teaching, research, and service at the University of Arkansas (1965–present) and has collaborated with numerous scientists on the biology and systematics of whiptail lizards (genera *Aspidoscelis* and *Cnemidophorus*: Family Teiidae). His graduate students have completed theses and dissertations on a variety of amphibian and reptile species. (Photographed by Shilpa Iyer).

Note

Occurrence of the California Red-legged Frog in Reservoirs Despite Habitat Alterations and Non-native Predatory Fish

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Abstract.—Losses of declining species can be confounding. Determining the origins of these losses can come from direct observation, literature review, or may be information passed along from biologist to biologist. A frequently cited literature source related to the natural history of California Red-legged Frog (*Rana draytonii*) may have painted a picture that was not only inaccurate but also has led to the dismissal of numerous opportunities to survey for and document the presence of this threatened species. Our observations from three municipal reservoirs in northern California suggest that California Red-legged Frogs occupy this habitat despite the putative acceptance of the absence of this species in reservoirs and downstream areas where predatory fish are sympatric. This conclusion is supported by a previous report for a reservoir in southern California. We found California Red-legged Frogs syntopic with predatory fishes, both native and non-native, and their downstream habitat. We do not suggest that reservoir construction has no effect on California Red-legged Frogs. Rather, we contend that it is important to survey reservoirs and associated upstream and downstream habitat for this species when presence/absence surveys are considered.

Key Words.—downstream; persistence; occupy, predatory fishes; *Rana draytonii*; recolonization; syntopic

The decline of the California Red-legged Frog (*Rana draytonii*; Fig. 1), a federally listed Threatened species throughout its range in California (U.S. Fish and Wildlife Service [USFWS] 1996), has been associated with introduced non-native species, habitat modification or destruction, water-quality degradation, and other factors (Moyle 1973; Jennings and Hayes 1994; USFWS 1996; Gilliland 2010; Thomson et al. 2016). Among contributing factors associated with the decline of the species are impoundment structures and water management projects (USFWS 1996). Although few dams have been constructed within the range of the species since 1994, water management projects and their associated habitat modifications routinely occur and are widely believed to be hindering the recovery of this threatened frog (Jennings and Hayes 1994).

Previous work on this species suggested that municipal reservoirs with concomitant habitat alteration and associated exotic species may have confounding negative effects on California Red-legged Frog populations and population persistence (Hayes and Jennings 1986). It has also been reported that upstream and downstream dispersal of exotic species can disrupt the community dynamics of California Red-legged Frogs (USFWS 2002). We assembled observations of California Red-legged Frog conducted during unrelated surveys from widely disparate areas over a 25-y period and examined those observations *post hoc* to determine the use of reservoirs by this species.

The dam on the Los Vaqueros Reservoir of the Contra Costa Water District located in Contra Costa County, California, was constructed in 1998 to improve domestic water quality and reliability for residents of central and eastern Contra Costa County (Fig. 2). Inundation of 566.6 ha of the upper Kellogg Creek watershed was complete by 1999 (https://www.ccwater.com/435/Los-Vaqueros-Project-History). The area of inundation included portions of Kellogg and Adobe Creeks, and approximately five minor tributaries and drainages. The reservoir was inundated from waters pumped through a
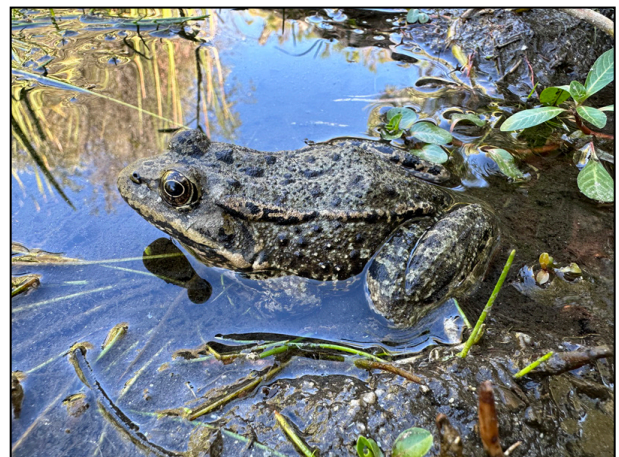

Figure 1. California Red-legged Frog (*Rana draytonii*), Wragg Ridge Preserve, Napa County, California. (Photographed by Jeff A. Alvarez).
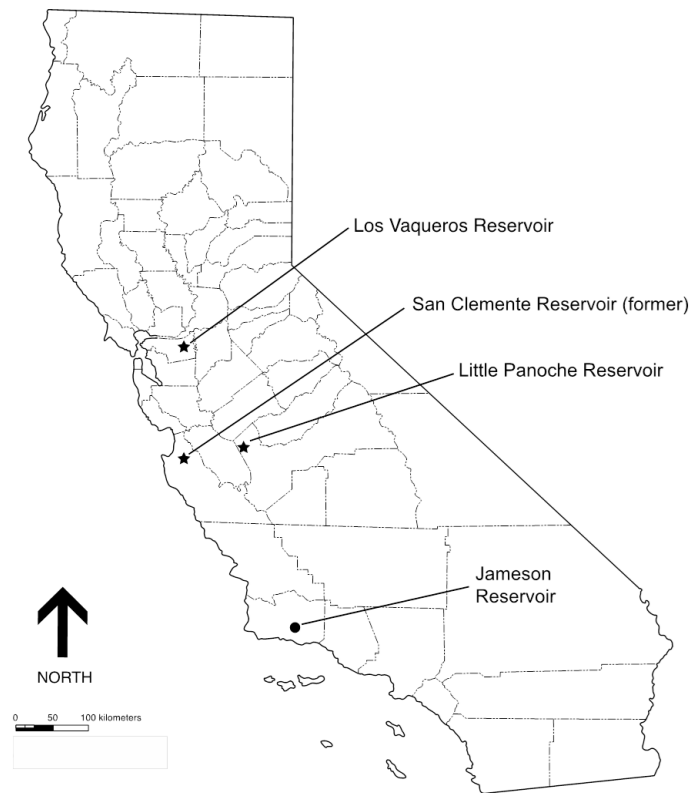
FIGURE 2. Locations of three reservoirs (star) that support California Red-legged Frogs (*Rana draytonii*) and predatory fishes in California. The location of the Jameson Reservoir (dot) included for reference.

pipeline connected to the Sacramento-San Joaquin River Delta (https://www.ccwater.com/435/Los-Vaqueros-Project-History). Shortly (one to two months) after pumping began, maintenance on the pumping station removed fish screens and allowed fish to freely enter the pipeline, which, thereafter, began to establish in the reservoir (pers. obs.). Among the many predatory species observed in the reservoir were Largemouth Bass (*Micropterus salmoides*) that exceed 4.5 kg and Striped Bass (*Morone saxattilis*) that exceed 18.1 kg, as well as Channel Catfish (*Ictularus punctatus*), White Catfish (*Ameiurus catus*), Brown Bullhead (*A. nebulosus*), Bluegill (*Lepomis macrochirus*), Green Sunfish (*L. cyanellus*), and numerous smaller, so-called bait fish (i.e., Mosquitofish, *Gambusia affinnis*, Inland Silverside, *Menidia beryllina*, Sacramento Blackfish, *Orthodon microlepidontus*, Threadfin Shad, *Dorosoma petenense*, Golden Shiner, *Notemigonus crysoleucus*, and Goldfish, *Carassius auratus*; Robert Nuzum, unpubl. report).

Following inundation, focused surveys for American Bullfrogs (*Lithobates catesbieanus*) and other non-native species were initiated by us. These surveys included both daytime and nighttime surveys during the late spring and summer months and included fish monitoring for the management of recreational fishing. Our surveys along the reservoir shoreline were expected to continue for the life of the reservoir, and occurred twice annually,

offering numerous opportunities to examine habitat around the reservoir.

Additionally, in October, November, and December 2023, we visited the Little Panoche Reservoir located in western Fresno County, California, which is a 16.6 ha reservoir that impounds water from the Little Panoche Creek drainage (Fig. 2). The reservoir was constructed in 1966 for the purpose of providing flood protection and agricultural and domestic water uses associated with the San Luis Canal, Delta-Mendota Canal, and other downstream developments (https://damsoftheworld.com/usa/california/little-panoche-detention-dam/). We conducted three nighttime Visual Encounter Surveys from both the shoreline and from kayaks on most of the shoreline edge.

We also conducted surveys at the San Clemente Reservoir, Carmel Valley, California. The dam at the San Clemente Reservoir was constructed in 1921 to supply water to the residents of western Monterey County and created a 576.1 ha reservoir (https://www.sanclementedamremoval.org/faqs). We conducted surveys for the presence of California Red-legged Frogs, including a 1,000 m downstream section of the Carmel River, in the summer of 1996 (Fig. 2). Surveyors walked the shoreline and conducted snorkel surveys (slowly moving along the shoreline edge and river channel) through all available habitats. We also included

observations reported by the USFWS (2002) from the Jameson Reservoir in Santa Barbara County. The Juncal Dam at this site was completed in 1930, creating a 55.8 ha reservoir, which provides drinking water to regional residents (https://montecitowater.com/our-water/water-sources-supply/surface-supplies/).

At the Los Vaqueros Reservoir, we made 40 survey visits that resulted in observations of 11–43 California Red-legged Frogs per survey. We found frogs in both adult and juvenile life stages in and around the reservoir every year during 20 y of surveys, 1998 to 2018. California Red-legged Frogs were typically observed within an extensive zone of interface between the remaining portions of Kellogg and Adobe Creeks, as well as in numerous other waterways that flow into the reservoir. As the waterways that flowed into the reservoir receded, we found California Red-legged Frogs occupying four of the five upstream reaches of drainages with fluctuating water levels, but where aquatic refuge habitat (i.e., upstream reaches of the arms of the reservoir) was maintained (e.g., shallow waters of the creek/reservoir delta). An incidental observation occurred in 2012 when we observed two California Red-legged Frogs in a small patch of partially inundated willow (*Salix* sp.) trees located approximately 12 m from shore and 300 m from the delta between the dry creek bed and reservoir inundation (i.e., well within the reservoir). We never observed California Red-legged Frogs in the waterways that flowed into the reservoir, which we also surveyed contemporaneously with reservoir surveys. This observation, in summer 2012 (14 y following inundation), was unanticipated, because the reservoir supported numerous predatory fish.

At the Little Panoche Reservoir, we found 182 juvenile and adult California Red-legged Frogs in the reservoir during the October 2023 surveys, and 56 juvenile and adult frogs in November and December 2023. Numerous predatory fish species were also present in the reservoir and upstream in Little Panoche Creek. These included non-native Black Bullhead (*Ameiurus melas*), Brown Bullhead, Channel Catfish, White Catfish, Largemouth Bass, Redear Sunfish (*L. microlophus*), Green Sunfish, Bluegill, Warmouth (*L. gulosus*), White Crappie (*Pomoxis annularis*), Black Crappie (*P. nigromaculatus*), as well as the native Sacramento Pikeminnow (*Ptychocheilus grandis*). In the downstream portions of the Little Panoche Reservoir, we also observed adult California Red-legged Frogs, and at least 47 juvenile frogs and one larva in downstream pools, 200 m downstream of the dam during a daytime Visual Encounter Survey, in July 2023.

At our third site, we found two adult California Red-legged Frogs among numerous adult and juvenile American Bullfrogs within emergent vegetation at the interface between San Clemente Creek and the San Clemente Reservoir. This observation coincided with predatory fish movements into the waterbody, as the native South/Central Coast Steelhead (*Oncorhynchus mykiss irideus*) annually moved up the creek into the reservoir (via a fish ladder) before migrating into the upper arms of the reservoir seeking spawning habitat (Moyle 2002; McGinnis 2006). The seasonal movement of steelhead into this coastal stream coincides with the spawning period of California Red-legged Frogs, which suggests that the two species are at least temporally sympatric (McGinnis 1984; Stebbins and McGinnis 2013). Subsequently, the San Clemente Dam was removed in 2015, which restored to conditions believed similar to pre-dam construction on the Carmel River (https://www.sanclementedamremoval.org). California Red-legged Frogs were present in the watershed and immediately recolonized the former reservoir area within the Carmel River (Gretchen Padgett-Flohr, pers. comm.). Even though we only surveyed this site in 1996, the observation of frogs first reported in 2015 suggests their persistence in the watershed. More recent observations of breeding California Red-legged Frogs in 2024 confirm occupation and population persistence at this site (Dawn Reis, unpubl. report). The USFWS (2002) also reported that California Red-legged Frogs were in Jameson Reservoir on the Santa Ynez River, Santa Barbara County, California. Fishes in the Jameson Reservoir include Largemouth Bass and native Rainbow Trout (*O. mykiss*), each of which is known or suspected of feeding upon California Red-legged Frogs (USFWS 1996).

We acknowledge that the surveys we conducted did not include monitoring of specific individual frogs. Therefore, we do not know if our repeated observations at a site (i.e., Los Vaqueros and Little Panoche Reservoirs) were transient individuals or the same individuals persisting in the habitat. Further, our surveys did not target and were not intended to detect egg masses or larvae. This would be important to investigate to determine if reproduction is occurring at the sites we surveyed, although reproduction must have been occurring at the Los Vaqueros Reservoir because we found frogs annually from 1998 to 2018.

Predatory fish are considered a significant threat to California Red-legged Frogs (USFWS 1996; Alvarez et al. 2002). Nonetheless, despite the presence of predatory fish that are known, believed, or likely to eat frogs, California Red-legged Frog have been found in these four reservoirs. Predatory fish-populated reservoirs that offer the appropriate microhabitat (i.e., high levels of habitat complexity) for refuge and dispersal seem to support California Red-legged Frog adults and juveniles over long periods.

It has been reported and widely cited that California Red-legged Frogs are generally extirpated from the drainages downstream of reservoirs within 1–5 y of impoundment (Hayes and Jennings 1988; Jennings and Hayes 1994; USFWS 2002). We, however, did not find this to be true at the sites we examined. For example, at the Los Vaqueros Reservoir, a large population of California Red-legged Frogs (approximately 7,000 to

10,000 individuals) have been extant within Kellogg Creek watershed for the 26-y life of the reservoir impoundment (USFWS 2002). It should be noted that a robust and effective control program for American Bullfrogs, as well as a high level of habitat complexity, may facilitate the persistence of this population in Kellogg Creek, downstream of the reservoir (Alvarez and Wilcox 2021; Alvarez and Wilcox, in press). We speculate that in the absence of American Bullfrog control (i.e., in the presence of bullfrogs), and presence of predatory fishes there might well have been a synergistic negative impact on the viability of California Red-legged Frogs in these drainages.

Our observations were similar in the downstream portions of the Little Panoche Reservoir where we recorded adult, juvenile, and larvae California Red-legged Frogs. These downstream observations of the presence of California Red-legged Frogs were in contrast to that reported by Jennings and Hayes (1994) and USFWS (2002). We also note that the decrease in numbers of observations from October to December at the Little Panoche Reservoir is likely due to the Allaback Effect (Allaback et al. 2010), whereby juveniles leave aquatic sites *en masse* during fall rain events, a commonly observed behavior for this species.

Our observations in the Los Vaqueros, Little Panoche, and San Clemente Reservoirs, and reports for the Jameson Reservoir (USFWS 2002) suggest that habitat use by California Red-legged Frogs, syntopic with predatory fishes over long periods of time (i.e., up to 26 y) is possible. Further, this may occur in the presence of certain anthropogenic structures, large-scale habitat modifications, continuous water-level fluctuations, as well as recreational fishing activity. We do not suggest that reservoir construction has no effect on California Red-legged Frog populations, or that this frog can routinely co-exist with, or reproduce, in the presence of predatory fishes and/or other significant habitat modification or degradation. Rather, we contend that reservoirs, and associated upstream and downstream habitat, even when predatory fish use these water bodies, should be considered viable dispersal and refuge habitat if California Red-legged Frogs are extant in the general area (Alvarez et al. 2002). All such habitats should be thoroughly surveyed to determine the potential presence of California Red-legged Frogs and how this declining species exploits various microhabitats within these types of reservoirs.

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SUSPECTED PREDATION BY COMMON RAVENS (*CORVUS CORAX*) ON KANGAROO RATS (*DIPODOMYS* SPP.) IN THE SAN JOAQUIN DESERT

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Abstract.—The Common Raven (*Corvus corax*) is a large-sized passerine that occurs in a variety of land types, including grasslands, desert scrub, and forests, ranging throughout a large portion of North America. Ravens are versatile opportunistic scavengers and predators that prey on a variety of species such as rodents, birds, and reptiles, as well as invertebrates and regularly consume anthropogenic food items. Herein, we report two raven suspected predation events on kangaroo rats (*Dipodomys* spp.), a nocturnal rodent.

Key Words.—Corvidae; diet; foraging; grassland; predation; prey.

The Common Raven (*Corvus corax*) is a large-sized passerine ranging throughout a large portion of North America. Ravens are habitat generalists that occupy and breed within a variety of land types within California, including grasslands, desert scrub, forests, and foothill regions (Boarman and Heinrich 1999). They are scavengers and formidable opportunistic predators that take a variety of vertebrate prey, including mammals, birds, and reptiles, and regularly consume eggs, invertebrates, and anthropogenic food items (Temple 1974; Engel and Young 1989; Camp et al 1993). Small mammals make up a significant portion of their diet. In a regurgitated pellet study conducted in Oregon, 59% of the food items identified were small mammals, including voles (*Microtis*), mice (*Peromyscus*, *Paragnathus*, *Reithrodontomys*), kangaroo rats (*Dipodomys*), and gophers (*Thomomys*; Stiehl and Trautwein 1991). In another pellet analysis study conducted in the Mojave Desert, California, mammals were found in 76.5% of the pellets examined, with kangaroo rats alone occurring in 282 pellets (50.2%; Kristan et al. 2004). These studies focused on pellet analysis and did not report on observed predation events occurring on the landscape. Herein, we document two incidents of suspected predation of kangaroo rats by Common Ravens within a foothill non-native grassland community in the San Joaquin Desert of western Kern County, California (approximately 18 km north of Blackwells Corner).

During the evening of 13 November 2024, just prior to sunset (1653; air temperature 15° C), we observed a solitary Common Raven flying west in front of us with a large item in its beak. We were able to identify the item as a kangaroo rat through binoculars due to its large hind feet and long, dangling tail. The following morning, 14 November, just after sunrise (0636; air temperature 10.5° C), we observed another raven transporting a kangaroo rat in its beak flying north in front of us with a second raven nearby in possible pursuit. Again, with binoculars, we were able to observe the kangaroo rat in the beak of the raven; this time we detected kicking motions as the kangaroo rat struggled while being held by the raven. The raven held the kangaroo rat by the midbody immediately behind the forelimbs. During these two observations, both ravens flew into the grassland fields and disappeared upon landing, presumably to consume their prey item. Two species of kangaroo rats occur in this area: the Giant Kangaroo Rat (*Dipodomys ingens*) and the Heermann's Kangaroo Rat (*D. heermanni*; Kelt 1988; Williams and Kilburn 1991). Although the Giant is larger than the Heermann's Kangaroo Rat, we could not identify the prey to species, especially at the distances we were making our observations, because they have the same body shape and markings (Williams et al. 1993).

Common Ravens, due to their generalist feeding habits, have the potential to impact special-status species, especially localized populations. For example, they have been documented predating on the Mojave Desert Tortoise (*Gopherus agassizii*; Camp et al. 1993; Knight et al. 1993) and the Western Burrowing Owl (*Athene cunicularia*; Clark 2017). Impacts to Giant Kangaroo Rats, a federally and state-listed Endangered species (USFWS 1998), are unknown, but predatory pressures by ravens on Giant Kangaroo Rats can potentially be a concern. Ravens are primarily active diurnally and kangaroo rats are predominantly nocturnal. Our observations occurred during crepuscular time periods (around sunrise and sunset) and may indicate that ravens have learned to exploit additional prey resources by extending their opportunistic hunting efforts into these time periods, especially during the late fall and winter months when day length is shorter.

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CAMERON A. REID grew up spending time outdoors through his involvement in Boy Scouts of America. Those experiences left him with a great respect for nature. Having lived in Fresno his whole life, he was able to spend considerable time in the adjacent Sierra Nevada and on the Central Coast. He completed a B.S. in Biology at California State University, Fresno, in 2022. He has spent the last 2 y working professionally at Colibri Ecological Consulting, LLC, as a Wildlife Biologist with a particular interest in San Joaquin Valley species. (Photographed by Cameron Reid).

PREDATION ATTEMPTS BY THE WESTERN YELLOW-BELLIED RACER (*COLUBER CONSTRICTOR MORMON*) ON THE PLATEAU STRIPED WHIPTAIL (*ASPIDOSCELIS VELOX*)

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Abstract.—Predator-prey interactions are typically infrequently observed. However, we report on observations of three predation attempts by the native Western Yellow-bellied Racer *(Coluber constrictor mormon)* on the introduced Plateau Striped Whiptail *(Aspidoscelis velox)* in Cove Palisades State Park, Jefferson County, Oregon. Triploid parthenogenetic *A. velox*, which is naturally distributed in parts of Arizona, Colorado, New Mexico, and Utah, has become established in the state park, where predator-prey interactions between these species are likely frequent occurrences. Our observations indicate that, behaviorally, *A. velox* often eludes attacks from *C. c. mormon*.

Key Words.—Oregon; predation; colubrid snakes; parthenogenetic; whiptail lizards.

Predation and successful foraging are vital to the survival of an animal; however, direct observations of such events are rarely witnessed first-hand (i.e., Wilcox and Sibanda 2023). Nevertheless, predation attempts and successes must occur quite frequently for such predatory organisms to survive. We report three predation attempts by the native Western Yellow-bellied Racer (*Coluber constrictor mormon*) on the introduced Plateau Striped Whiptail (*Aspidoscelis velox*) in the Cove Palisades State Park, Jefferson County, Oregon.

This predator species is an actively foraging and opportunistic snake that occurs over most of North and Central America from southern Canada to Guatemala (Crother 2017). It inhabits a wide variety of habitats including grasslands, woodlands, meadows, and sub-steppe desert (Shewchuk and Austin 2001). This ophidian consumes a variety of invertebrate and vertebrate prey including grasshoppers, crickets, small mammals, snakes, lizards, and amphibians (Halstead et al. 2008). Halstead et al. (2018) found that a population of this snake in Florida commonly preyed upon the actively forging Six-lined Racerunner (*Aspidoscelis sexlineatus*).

Triploid parthenogenetic *A. velox* is also an actively foraging lizard that naturally occurs in parts of Arizona, Colorado, New Mexico, and Utah (Cole et al. 2019), but has been introduced to Cove Palisade State Park, Jefferson County, Oregon (Kusaka et al. 2024). It was suggested that the species was introduced to north-central Oregon sometime before 1970 (Storm et al. 1995). *Aspidoscelis velox* occupies a wide range of habitats including desert scrub, grasslands, and Pinon-Juniper Woodlands (Persons and Wright 2009). In Oregon, the lizard inhabits rocky juniper-grown areas on the west side of Lake Billy Chinook in Cove Palisades State Park (Storm et al. 1995).

We observed *C. c. mormon* launch attacks on *A. velox* in three separate occasions in Cove Palisades State Park. As part of another research project to assess the status of *A. velox* in the park, two observers surveyed in the park from about 0900–1400 during 17–18 July 2023 in the vicinity of the Deschutes Campground and the Tam-a-láu Trail (Kusaka et al. 2024). On 17 July 2023, while surveying the area close to the campground right before the Tam-a-láu Trail head, we observed three unsuccessful predation attempts on *A. velox* by *C. c. mormon*.

The first predation attempt occurred at 1000, when observers spotted a *C. c. mormon* moving through the vegetation. Observers followed behind to accurately identify the snake, during which they observed the snake raised its head slightly, sped up, and dart forward after a passing *A. velox*. The *C. c. mormon* was unsuccessful at catching or biting the lizard, but continued to chase after the lizard, so observers continued to follow the interaction and observe if a successful predation event would occur. Shortly after, the *A. velox* sprinted away and was lost from sight behind vegetation and under a fence sectioning off part of the Deschutes campground. The *C. c. mormon* continued in the general direction but slowed pursuit until it seemed to stop following the lizard and moved in a different direction. The predation attempt was over quickly, but observers followed for an extra 2 min to see if more predation attempts would occur.

At 1005 while moving back to the initial location where the survey was interrupted by the first observation, observers noticed another *C. c. mormon* chasing another *A. velox* through the vegetation. This *A. velox* was darting

swiftly from one patch of vegetation to another and through grasses with the *C. c. mormon* following close behind. When the snake got close (ca. 0.5 m), it darted forward trying to strike at the *A. velox*, which sprinted several meters away and was lost from sight behind more vegetation. This observation was also over quickly, but we did not follow either lizard or snake further as we would not have been able to confirm if any new *A. velox* spotted was the same lizard being initially chased. We are confident that these two predation attempts involved different pairs of *C. c. mormon* and *A. velox* due to the short time frame and distance ($\geq$ 20 m) between observations, and with the animals travelling in opposite directions from each other.

The final observation occurred at 1036 and was very similar to the previous predation attempts in which the *C. c. mormon* seemed to be following the *A. velox*, only for the lizard to successfully outrun the snake. Throughout these seemingly unsuccessful predation attempts, we tried to stay a minimum of 3–5 m away from the animals with the distance varying due to the density of the vegetation. All three predation attempts occurred between 1000 and 1036 just south of the Tam-a-láu trailhead near the B-loop campground (Fig. 1) in a landscape dominated by grasses and big sagebrush (*Artemisia tridentata*).

It was likely that only one or a few individual *A. velox* were initially introduced to the Cove Palisades State Park, but as a parthenogenetic species they have continued to persist and spread (Kusaka et al. 2024). Given the multiple predation attempts we observed in a relatively short time period (three observations over 2 d of surveying), these types of predator-prey interactions between *C. c. mormon* and *A. velox* may be quite common. The introduction of *A. velox* to Cove Palisades State Park, adds another potential prey to the opportunistic feeding habits of *C. c. mormon* and could be or become an important food source as *A. velox* continues to spread in Oregon.

Figure 1. Map of the Deschutes campground, relative to Lake Billy Chinook, and the Tam-a-láu Trail (orange), Cove Palisades State Park, Jefferson County, Oregon. The red polygons represent the flat areas of habitat surveyed where adult Plateau Striped Whiptails (*Aspidoscelis velox*) were observed. Predation observations occurred in the middle polygon (3rd from the top) by the start of the Tam-a-láu Trail. The location of Cove Palisades State Park and the Deschutes campground in Oregon (red dot) can be seen in the map in the bottom right corner.

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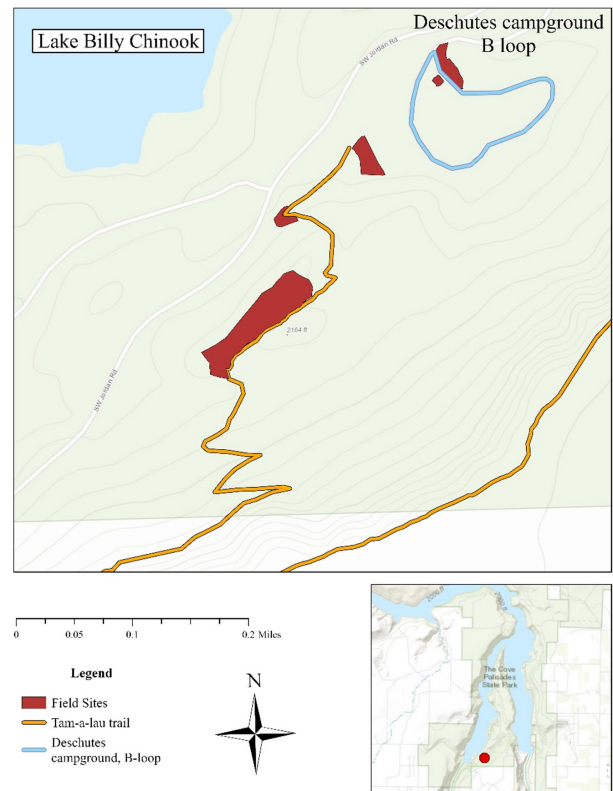
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CARINA M. KUSAKA is a Graduate Research Assistant at Oregon State University in Corvallis, Oregon, USA. Her current research focuses on spatial analyses in the breeding habitat of the Tufted Puffin (*Fratercula cirrhata*). She received a B.S. in Fish, Wildlife, and Conservation Biology from Colorado State University, Fort Collins, Colorado, USA, where she worked with Erell Institute to study the behavioral and movement ecology of reptiles. Carina is primarily interested in understanding the effects of climate change on species of greatest conservation need and using art to help foster excitement and education in communities for their local ecosystems. (Photographed by Kenny Kusaka).

KAERA L. UTSUMI is a Behavioral Ecologist specifically interested in reptile behavior. She is currently a Research Assistant at Erell Institute, Lawrence, Kansas, and has worked with the organization on multiple research projects in several countries engaging women and underrepresented groups in science. Some of the lizards Kaera has studied include Atacamen Pacific Iguana (*Microlophus atacamensis*), Colorado Checkered Whiptail (*Aspidoscelis neotesselatus*), and Namib Web-footed Gecko (*Pachydactylus rangei*). When she is not working on research projects, Kaera enjoys reading, crocheting, and playing basketball. (Photographed by Maria Eifler).

JAMES M. WALKER is Professor of Biological Sciences, University of Arkansas, Fayetteville, Arkansas, USA. Since earning B.S. and M.S. degrees from Louisiana Tech University, Ruston, USA, and Ph.D. from the University of Colorado, Boulder, USA, he has engaged in teaching, research, and service at the University of Arkansas (1965–present) and has collaborated with numerous scientists on the biology and systematics of whiptail lizards (genera *Aspidoscelis* and *Cnemidophorus*: Family Teiidae). His graduate students have completed theses and dissertations on a variety of amphibian and reptile species. (Photographed by Shilpa Iyer).

2024 Annual Meeting Review
71ˢᵗ Annual Meeting
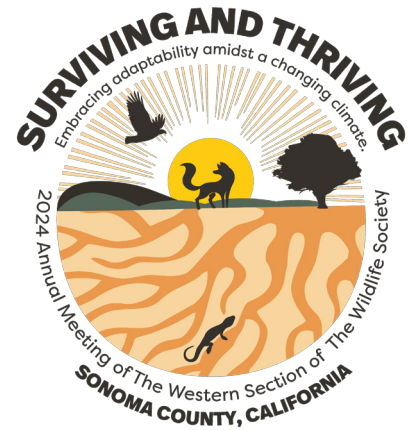
Program Chair: Brooke Langle, SWCA

Attendance: 753 participants including 106 students and 40 Early Career Professionals.

Plenary Theme: Surviving and Thriving: Embracing our Adaptability Amidst a Changing Climate

Plenary Speakers:

- **Wendy Millet**, TomKat Ranch Educational Foundation
- **Justin Winters**, One Earth

Keynote Address - California's Estuaries in the Face of Climate Change

- **Dr. Christine Whitcraft**, California State University Long Beach
- **Dr. John Takekawa**, Suisun Resource Conservation District

Capstone Presentation:

- **Kim Stanley Robinson,** Author

2024 Pre-Conference Workshops:

- Snake Symposium
- Introduction to R Workshop
- Sonoma Field Trips

Student Presentation Awards

Oral Presentation Awards

1ˢᵗ: Julia Owens-Ramos - Use of Genomic Tools to Investigate the Evolutionary History of Island Spotted Skunks

2ⁿᵈ: Katie Elder - Density-Dependent Habitat Selection and Home Range Sizes in a Recovering Population of Island Foxes (Urocyon littoralis) on Santa Rosa Island, California

3ʳᵈ: Katie Saenger - Movement of weaned northern elephant seal pups during their first at sea 4 gene migration

Poster Presentation Awards

1ˢᵗ: Isabelle Smits - Behavioral Responses to Anti Predator Training in Head-Started Juvenile Desert Tortoises

1ˢᵗ: Christopher Collier - Major Carnivore Use of a Post Fire Landscape: Impacts of the Dixie Fire in Lassen and Plumas National Forest, California

2ⁿᵈ: Cassandra Rodriguez - Genomic Sequencing to Compare Pedigree-Based and Genomic Inbreeding Measures for a Small, Isolated Mountain Lion Population

3ʳᵈ: Alissa Cox - Winter Diet of the Mexican Free-Tailed Bat (*Tadarida brasiliensis mexicana*) in Northern California

2024 Awards Bestowed:

- The Raymond F. Dasmann Award for the Professional of the Year went to Tiana Williams-Clausen
- The Conservationist of the Year Award went to the California Bumble Bee Atlas
- The Chapter of the Year Award went to the Sacramento-Shasta Chapter
- The James D. Yoakum for Outstanding Service and Commitment to The Western Section of The Wildlife Society went to Dr. Jeff Lincer
- The Barrett A. Garrison Mentor of the Year Award went to Ivan Parr

2024 Western Section Workshops:

- **Vernal Pool Branchiopods**, January 22-27, 2024, Davis/Sacramento, CA
- **Hawaii Forest Birds**, February 17-20, 2024 in Hawaii
- **Mohave Ground Squirrel Workshop**, March 22-24, 2024 in Ridgecrest, CA
- **The Bumble Bee Field Course**, July 19-21, 2024 in Lake County, CA
- **Introduction to Tree Voles**, July 26-28, 2024 in Corvallis, OR
- **Women+ of Wildlife Retreat**, September 27-29, 2024 in Santa Ynez, CA
- **Western Monarch Overwintering Ecology & Habitat**, November 13-14, 2024 in San Luis Obispo County

2024 Western Section Membership

Retired: 54	Student: 240	**Total: 1310**
Regular: 803	New Professional: 146	
Lifetime: 53	Supporting: 7	**(does not include Honorary)**
Contributing: 7	Honorary: 46	

TWS Western Section Board Members — 2024