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NOTES

COOPERATIVE DEFENSE OF A NEST SITE BY SWAINSON'S HAWKS (*BUTEO SWAINSONI*)

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***Abstract*.—Here I report on an incident in which a large group of Swainson's Hawks (*Buteo swainsoni*) collectively drove a Golden Eagle (*Aquila chrysaetos*) away from a nest site.**

Key Words.—California; predation; raptors

On 12 July 2017 at 1320 h, in Madera County, California, I observed a large group of Swainson's Hawks (*Buteo swainsoni*) act together to drive a Golden Eagle (*Aquila chrysaetos*) away from a nest site. I observed the incident during a follow up visit to a Swainson's Hawk nesting site I had detected earlier and used 8×42 binoculars to assist my observations. When I arrived at my observation point about 300 m south of the nest tree, two Swainson's Hawks were engaged in harassing a Golden Eagle. The two hawks repeatedly stooped over the eagle. After about 10 min, the birds moved out of sight to the east. A few minutes later the birds appeared north of the nest and then moved out of sight. I drove 2 km toward the north and again observed the birds. From the new location I observed three Swainson's Hawks harassing the eagle. After a few seconds, I realized a kettle of 14 Swainson's Hawks had formed over the eagle. Periodically an additional hawk would descend from the kettle and harass the eagle. By 1350, the eagle had flown out of sight, but several Swainson's Hawks stayed within 1 km of the nest tree. These birds engaged in an aerial display, including short stoops and slower controlled descents for several minutes before dispersing.

Mobbing is a well-known defensive tactic among closely nesting birds. Arroyo et al. (2001) reported mobbing by a colonial nesting raptor, the Montagu's Harrier (*Circus pygargus*), in which up to eight harriers from nearby nests were recruited to assist in mobbing. Up to 50 Marsh Harriers (*Circus aeruginosus*) participated in mobbing near a colonial roost (Sternalski and Bretagnolle 2010). The literature provides records of fewer Swainson's Hawks cooperating in nest defense. Fitzner (1980) recorded two incidents of a non-breeding Swainson's Hawk sharing nesting territories and even assisting in nest defense with the paired birds. Cash (1989) suggests polygamous behavior may have been involved in a similar case that he observed. Thurow and White (1983) documented cases in which Swainson's Hawks nested near Ferruginous Hawks (*Buteo regalis*) and the associating pairs acted cooperatively in defense of each other's nests against both terrestrial intruders and Golden Eagles. The incident I observed was likely a form of mobbing, with loosely associated Swainson's Hawks from nearby territories participating. As far as I could determine, no one has previously documented a large group of Swainson's Hawks acting together to defend a nest site.

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UPDATED DISTRIBUTIONS OF THREE SPECIES OF YELLOW BAT (*DASYPTERUS*) IN TEXAS BASED ON SPECIMEN RECORDS

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***Abstract*.—Updating species ranges and documenting range extensions with new county records helps monitor the status of populations, provides insight into life-history traits, and informs conservation decisions. Three species of yellow bats (*Dasypterus* spp.) reach the limits of their distribution in Texas and have been documented to be expanding their previously known range. Here, we updated the distribution maps of the three species of yellow bats (*D. ega*, *D. intermedius*, and *D. xanthinus*) in Texas using vouchered specimens. We confirmed species identities using morphological measurements and, in some cases, sequence from a fragment of the mitochondrial cytochrome *b* gene (*Cytb*). We documented eight county records for *D. ega*, eight county records for *D. intermedius*, and one county record for *D. xanthinus* in Texas. Updated distribution maps produced using county records help refine our understanding of the natural history of these species.**

Key Words.—Chiroptera; cytochrome *b*; *Lasiurus*; range extension

INTRODUCTION

Yellow bats are medium-large vespertilionid bats, are insectivorous, roost in trees, and are found in North, Central, and South America. Three species of yellow bats occur in the U.S.: Southern Yellow Bats (*Dasypterus ega*), Northern Yellow Bats (*D. intermedius*), and Western Yellow Bats (*D. xanthinus*). Yellow bats were formerly included in the genus *Lasiurus*, until reassigned to *Dasypterus* by Baird et al. (2015) based on genetic distance and divergence times between yellow bats and other *Lasiurus* bats. Additionally, Western Yellow Bats were originally named as a subspecies of Southern Yellow Bats (*L. ega xanthinus*) until elevated to species level in 1988 (Baker et al. 1988).

Migration in these species may be facultative or absent, as individuals from portions of their respective ranges have been demonstrated to be present year-round (Baker et al. 1971; Zabriskie et al. 2019). Litter size in Western Yellow Bats is typically two, whereas in Southern and Northern Yellow Bats the litter size is 2–4 (Ammerman et al. 2012). Yellow bats roost primarily in the dried fronds of palm trees such as Mexican Fan Palms (*Washingtonia robusta*) and Texas Sabal Palms (*Sabal mexicana*) in Texas (Chapman and Chapman 1990; Jimenez 2016). Northern Yellow Bats also roost in Spanish Moss (*Tillandsia usneoides*) associated with hardwood trees in the southeastern U.S. (Constantine 1958; Coleman et al. 2012), and Western Yellow Bats have been documented roosting in Dagger Yucca (*Yucca carnerosana*) in Big Bend National Park (Higginbotham et al. 2000) and deciduous trees, such as sycamore (*Platanus* spp.) and cottonwood (*Populus* spp.; Bond 1970).

Southern Yellow Bats are medium-sized (forearm length: 42–48 mm), have yellow-brown pelage, and are distributed from south Texas to Argentina (Esbérard and Moreira 2006; Ammerman et al. 2012). Northern Yellow Bats are the largest of the yellow bats (forearm length: 45–56 mm) with yellow-brown pelage and are distributed from South Carolina in the U.S. to northern Nicaragua, primarily inhabiting coastal regions (Webster et al. 1980; Ammerman et al. 2012). Western Yellow Bats are of a similar size to Southern Yellow Bats (forearm length: 43–47 mm), have pale yellow pelage, and occur in southwestern U.S. and northeastern Mexico (Ammerman et al. 2012). The three species are difficult to distinguish from one another morphologically, especially as juveniles, and require consideration of range as well as genetic techniques to confirm identification to species level (Baker et al. 1988; Morales and Bickham 1995; Tipps et al. 2011).

In the U.S., each of the three species of yellow bats reach limits of their distributions in Texas. Southern Yellow Bats meet the northern limit of their range in Texas and are historically only known from the southernmost counties of Texas. Texas is the westernmost limit of the range of Northern Yellow Bats, which are primarily a coastal species; in Texas their range follows the coast of the Gulf of Mexico. Western Yellow Bats were first documented in Texas in 1996 in Big Bend National Park, Brewster County (Higginbotham et al. 1999). Since this first record, the species has been documented in three additional Texas counties and appears to be restricted to western Texas, the easternmost limit of their range.

The range of Southern Yellow Bats has recently extended as reported by Demere et al. (2012) with an extension of the northern boundary into Fayette and Comal counties. Demere et al. (2012) also reported extension of Northern Yellow Bats into Webb, Frio, Uvalde, and Starr counties, representing extension of the western range boundary. Tipps et al. (2011) reported extension of Western Yellow Bats into El Paso County; however, these specimens tested positive for the rabies virus and therefore may have exhibited abnormal behavior. Additional specimens are important to confirm the presence of Western Yellow Bats in El Paso County. Range extension for Western Yellow Bats has also recently been reported in the Chihuahuan Desert Ecoregion of New Mexico and a new record from Las Cruces, New Mexico, represents a northeastward range extension of approximately 210 km in the state (Zabriskie et al. 2019). Yellow bats are all considered to be rare across their respective ranges and thus, much is unknown about their life-history traits and range limits. The objective of this study was to update the distribution of the three species of yellow bats in Texas based on current occurrence records.

Methods

Yellow bats that we examined were submitted to the Texas Department of State Health Services (DSHS) in Austin, Texas, between 2008 and 2019 for rabies testing. Of those submitted during this time period, approximately 760 yellow bats tested negative for the rabies virus, were frozen, and were added into a database that was used to identify new county records. We determined specimens to be new county records if we did not find a museum voucher specimen for a species from the county. We used distribution maps in Ammerman et al. (2012), lists of specimens examined on the website for Bats of Texas (Ammerman et al. 2012; www.batsoftexas.com), and new county record papers (Demere et al. 2012, Tipps et al. 2011) to determine records of interest. Some counties had only literature records or Texas DSHS records, indicating that a specimen was recorded in the county but no voucher specimen was identified. Texas DSHS records are generally comprised of specimens that were incinerated after being recorded in the database and thus no collections-based voucher specimen exists. Specimens were incinerated if they tested positive for rabies or if they were not deposited at another institution.

Voucher specimens were deposited at the Angelo State Natural History Collections (ASNHC) or the Natural Science Research Laboratory at Texas Tech University (NSRL-TTU) and were prepared as skull only, study skin, or fluid preserved specimens. Fluid specimens were preserved in 70% ethanol after fixation in 10% formalin. The county of collection and the date of collection of the specimen by Texas DSHS were recorded for each specimen. More specific collection data, such as specific locality or information about collectors, was unavailable due to data privacy restrictions set by the Health Insurance Portability and Accountability Act of 1996 (HIPAA). Age (adult or juvenile) and sex were determined and recorded for each specimen and standard morphological measurements (total length, tail length, hindfoot length, ear length, length of the tragus, and forearm length) were taken and recorded. We used measurements and other key characteristics as described in Ammerman et al. (2012) to confirm species identity. Tissue samples (heart, kidney, liver, spleen, lung, and muscle) were taken when possible, frozen, and deposited at the ASNHC (issued an ASK number).

Due to the morphological similarities of the species of *Dasypterus*, we confirmed the species identity of some specimens using DNA sequence data from the mitochondrial cytochrome *b* (*Cytb*) gene, commonly used for species identification by DNA barcoding in mammals (Kocher et al. 1989; Irwin et al. 1991). We extracted DNA from frozen tissues (heart, kidney, or liver) according to the DNeasy Blood and Tissue Kit (Qiagen Inc., Valencia, California) protocols. We visually assessed the quality of the DNA using gel electrophoresis (0.8% agarose in 1× sodium borate solution) and quantified for DNA yield and purity using a NanoDrop Lite (Thermo Fisher Scientific Inc., Waltham, Massachusetts) spectrophotometer or a Qubit 1.0 fluorometer (Invitrogen Corp., Carlsbad, California). We amplified *Cytb* using polymerase chain reaction (PCR) primer sets MVZ05/MVZ04 (Smith and Patton 1991) or LGL765/LGL766 (Bickham et al. 1995). Amplifications were done using 1× standard Taq reaction buffer (New England BioLabs, Ipswich, Massachusetts), 1 unit of Taq polymerase (New England BioLabs), 2.0 mM $MgCl_2$ (New England BioLabs), 0.2 mM of each dinucleoside triphosphate (Thermo Fisher Scientific Inc.), 0.16 µM of each forward and reverse primer, 50–500 ng of template DNA, and RNase free water as needed to meet a final reaction volume of 12.5 µL.

The thermal profile for the MVZ05/MVZ04 primer set began with a denaturing temperature of 94° C for 2 min, followed by 39 cycles of 94° C for 1 min, 50° C or 48° C for 1 min, 72° C for 1 min, followed by a final extension of 72° C for 5 min. The thermal profile for the LGL765/LGL766 primer set had an initial denaturing step of 94° C for 2 min, followed by 35 cycles of 92° C for 1 min, annealing at 52° C for 1 min, 72° C for 1 min, and a final extension of 72° C for 5 min. We verified product amplification with gel electrophoresis (0.8% or 1% agarose in 1× sodium borate solution). We then purified PCR products to prepare for sequencing with ExoSAP-IT PCR Product Cleaning Reagent (Thermo Fisher Sci.) following the protocol of the manufacturer. For the MVZ05/MVZ04 primer set, we sequenced the purified samples following the protocol of the manufacturer with the GenomeLab DTCS-Quick Start Kit in a Beckman Coulter CEQ 8000 Genetic Analysis

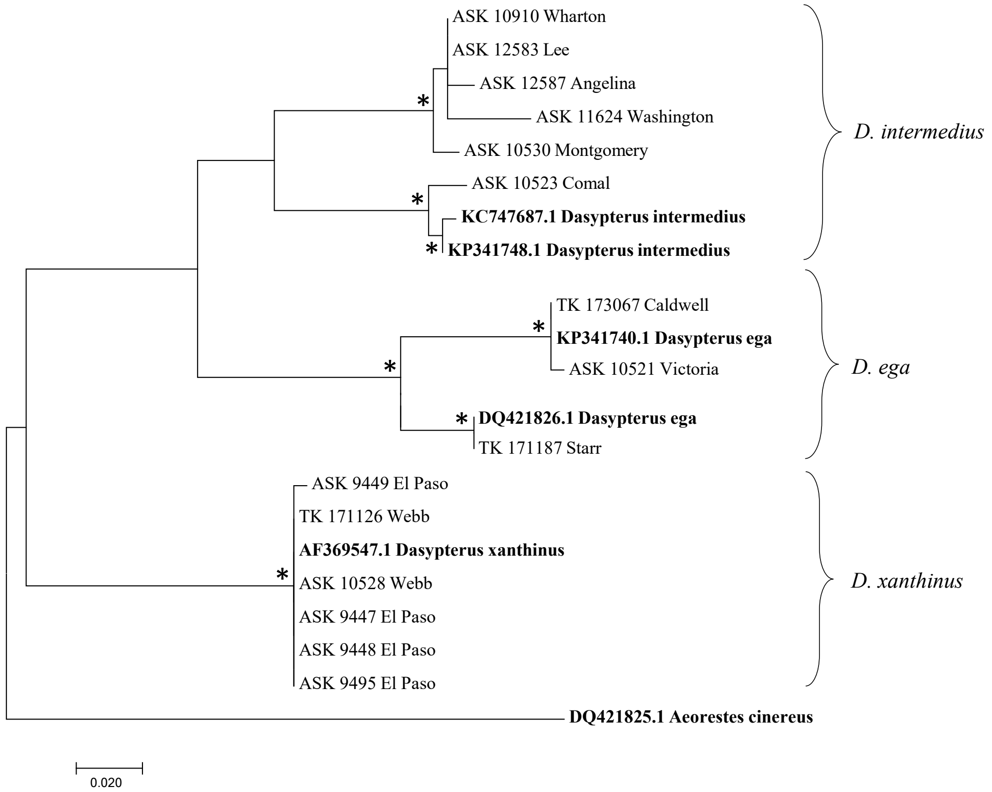

Figure 1. Neighbor-joining tree generated from 258 base-pair fragments of the cytochrome *b* gene from yellow bat specimens recovered from the Texas Department of State Health Services. Sample names are based on collection tissue numbers and county of origin. Reference sequences from Genbank are in bold and were included to identify clusters. The Hoary Bat (*Aeorestes cinereus*) was used as an outgroup. An asterisk (*) at nodes indicates high bootstrap support (> 90%).

System (Beckman Coulter Inc., Brea, California). We sent samples for Sanger sequencing, amplified using the LGL765/LGL766 primer set, to the Genomic Core Sequencing Lab at Texas A&M, Corpus Christi, Texas.

All sequences were edited and compared to reference sequences of the same species with Sequencher v. 5.1 (Genecodes Corp., Ann Arbor, Michigan). Consensus sequence data were then exported and aligned in MEGA 7 (Kumar et al. 2016) using the MUSCLE alignment tool. We analyzed aligned sequences and we constructed a neighbor-joining tree using a 258 base-pair fragment of *Cytb* to show the phylogenetic relationships between the *Dasypterus* species (Fig. 1). Nodal support was evaluated with 1,000 bootstrap replicates. We included reference sequences of known species of yellow bats from Genbank (GB) in the alignment (*D. ega*: DQ421826 and KP341740.1; *D. intermedius*: KC747687.1 and KP341748.1; *D. xanthinus*: AF369547.1). We also included a sequence from a Hoary Bat (*Aeorestes cinereus*) from GB (DQ421825.1) to be used as an outgroup. We confirmed specimen identification by the clustering pattern and by the similarity to reference *Cytb* sequences. Once we confirmed species identities, we created updated distribution maps with QGIS 3.4 (QGIS Development Team 2019) indicating new county records or those updated from Texas DSHS records.

RESULTS

Dasypterus ega (*Southern Yellow Bats*).—Southern Yellow Bats in the U.S. are known only from the southernmost counties of Texas. Here, we present eight new county records for Southern Yellow Bats (Fig. 2A) and extend the northern limit of their range into Travis and Montgomery counties in Texas. A specimen from San Patricio County represents an update from a Texas DSHS record to a collections-based specimen record.

Bandera County.—A male specimen (ASNHC17404, ASK11627) was received on 12 February 2014.

Caldwell County.—A male specimen (TTU114567, TK173067) was received on 9 June 2011. We confirmed the species identity of this specimen (Fig. 1) with *Cytb* (GB accession MN895072).

Hays County.—Two specimens were recovered from Hays County. A male specimen (ASNHC17403, ASK11626) was collected on 5 March 2014. A female specimen (ASNHC19953, ASK13748) was collected on 22 January 2018.

Montgomery County.—A specimen of undetermined sex (ASNHC16290) was received on 8 February 2013.

San Patricio County.—A female specimen (ASNHC18197, ASK12697) was collected on 24 October 2013.

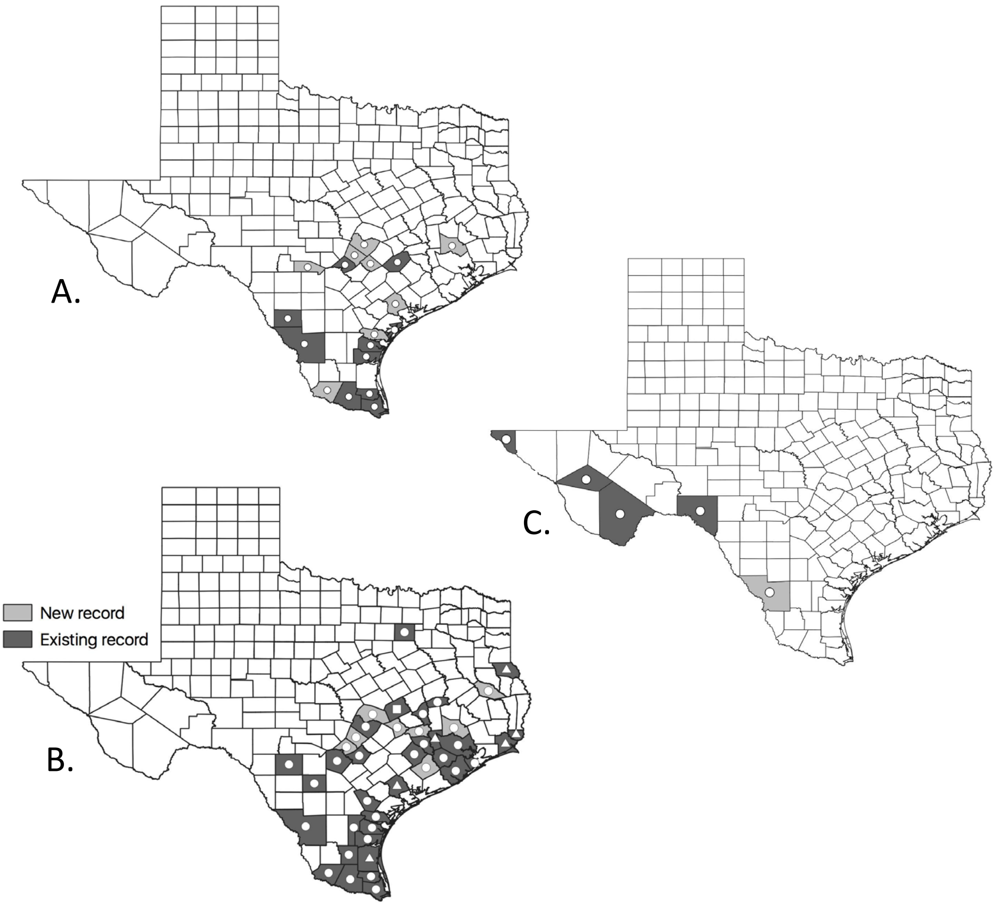

Figure 2. Updated distribution maps for (A) Southern Yellow Bats (*Dasypterus ega*), (B) Northern Yellow Bats (*D. intermedius*), and (C) Western Yellow Bats (*D. xanthinus*). New county records are shaded with light gray. Symbols are circles = specimen vouchers, squares = literature records, triangles = Texas Department of State Health Services records.

Starr County.—A female specimen (TTU113575, TK171187) was received on 11 July 2008. We confirmed the species identity (Fig. 1) with *Cytb* (GB MN895073).

Travis County.—A male specimen (ASNHC19335, ASK13746) was collected on 5 September 2018.

Victoria County.—A male specimen (ASNHC15835, ASK10521) was collected on 29 June 2012. We confirmed the species identity of this specimen (Fig. 1) with *Cytb* (GB MN895074).

Dasypterus intermedius (*Northern Yellow Bats*).—Northern Yellow Bats have been primarily collected in counties in Texas along the coast of the Gulf of Mexico, but recently appear to be expanding inland (Demere et al. 2012). We present eight new county records for Northern Yellow Bats (Fig. 2B), including new collections-based specimen records for two counties that had previously been considered Texas DSHS records (Montgomery and Wharton counties). These records are from counties within the known distribution of Northern Yellow Bats.

Angelina County.—A female specimen (ASNHC17877, ASK12587) was collected on 7 September 2016. We confirmed the species identity of this specimen (Fig. 1) with *Cytb* (GB MK876232).

Comal County.—A male specimen (ASNHC15836, ASK10523) was received on 11 September 2012. We confirmed the species identity of this specimen (Fig. 1) with *Cytb* (GB MN895076).

Hays County.—A female specimen (ASNHC19952, ASK13747) was received on 17 October 2018.

Lee County.—A male specimen (ASNHC17873, ASK12583) was collected on 6 October 2016. We confirmed the species identity of this specimen (Fig. 1) with *Cytb* (GB MK876234).

Montgomery County.—Two specimens were recovered from Montgomery County. A male specimen (ASNHC15863, ASK10530) was received on 17 July 2012. We confirmed the species identity this specimen (Fig. 1) with *Cytb* (GB MK876235). An additional male specimen (ASNHC17872, ASK12582) was collected on 15 September 2016.

Washington County.—A male specimen (ASNHC17401, ASK11624) was received on 7 March 2014. We confirmed the species identity of this specimen (Fig. 1) with *Cytb* (GB MK876252).

Wharton County.—A male specimen (ASNHC19059, ASK10910) was received on 23 June 2015. We confirmed the species identity of this specimen (Fig. 1) with *Cytb* (GB MK876253).

Williamson County.—A male specimen (ASNHC19300, ASK13849) was received on 13 October 2014.

Dasypterus xanthinus (*Western Yellow Bats*).— Since Western Yellow Bats were first documented in Texas, the species has been primarily reported in the Big Bend and Trans-Pecos regions and has only been collected in Texas in Brewster, Jeff Davis, Val Verde, and El Paso counties. Here, we report an additional county in which Western Yellow Bats have been collected (Fig. 2C). Specimens from Webb County represent the easternmost records of Western Yellow Bats in the U.S. This eastern extension increases the difficulty of identifying yellow bats to species level as the three species have all been reported in Webb County and their ranges likely overlap in other counties in southern Texas.

El Paso County.—Western Yellow Bats have been previously reported in El Paso County (Tipps et al. 2011); however, those specimens tested positive for rabies virus and may have exhibited abnormal behavior. Here, we report four additional records of Western Yellow Bats collected in El Paso County, all of which tested negative for rabies virus to corroborate previous records. A female specimen (ASNHC15826, ASK9495) was received on 30 July 2012. An additional female specimen (ASNHC16080, ASK9449) was collected on 04 September 2012. A male specimen (ASNHC16079, ASK9448) was collected on 21 September 2012. An additional male specimen (ASNHC16078, ASK 9447) was received on 26 September 2012. We confirmed the species identity for these four specimens (Fig. 1) with *Cytb* (GB MN895077, MN895078, MN895079, MN895080, respectively).

Webb County.—Two specimens were collected from Webb County. A female specimen (TTU113514, TK171126) was collected on 25 June 2009. We confirmed the species identity for this specimen with *Cytb* (GB MN895081). Additionally, a male specimen (ASNHC15800, ASK10528) was collected on 19 July 2012. We confirmed the identity of this specimen (Fig. 1) with *Cytb* (GB MN895082).

DISCUSSION

Specimens submitted to state health departments offer unique insight into the distributions of elusive species, especially in areas that are not sampled frequently with conventional collection methods (Yancy and Jones 1996; Tipps et al. 2011; Demere et al. 2012). Documenting shifts in species distributions with new county records provides valuable data to help monitor the status of populations and diseases, and gives insight into life-history traits such as habitat preferences and migration habits. It is also important to document that such changes as anthropogenic activity and climate change alter habitat suitability for many species. Though the three species of yellow bats that occur in the U.S. are listed as species of least concern on the Red List of the International Union for Conservation of Nature (IUCN), Southern Yellow Bats are listed as threatened in Texas (Texas Parks and Wildlife. 2019. Federal and State Listed Mammals in Texas. Available from https://tpwd.texas.gov [Accessed 22 November 2019]) and Western Yellow Bats are listed as threatened in New Mexico (Jones and Schmitt 1997).

The possible range expansion of yellow bats has been hypothesized to correspond to the introduction of ornamental palms in residential areas, as noted in Western Yellow Bats in southern California (Constantine 1998). Though the use of ornamental palms in landscaping can provide more roosting sites for yellow bats, the practice of pruning the dried leaves puts roosting bats at risk of losing roosts or getting injured when the leaves are removed (Mirowsky 1997; Zabriskie et al. 2019). Other major threats that affect these species are pesticide use (Clark et al. 1978; Clark 2001) and collisions with wind turbines (Arnett et al. 2016). As anthropogenic activity continues to alter landscapes and change habitat suitability for other organisms, documenting range shifts and monitoring population trends will be necessary for effective conservation assessment of bat species.

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RECONNECTING SMALL MAMMAL POPULATIONS IN THE CASCADE RANGE ACROSS AN INTERSTATE HIGHWAY: AN EARLY LOOK AT USE OF A WILDLIFE CROSSING STRUCTURE

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Abstract.—**Expanding transportation corridors have fragmented ecosystems throughout the world, restricting the movement of organisms or acting as complete connectivity barriers. Wildlife crossing structures (WCS) can increase the permeability of roads by allowing animals to move safely between habitats. Small mammals are especially vulnerable to the effects of reduced connectivity because of their limited mobility; however, researchers have only recently begun to evaluate their use of WCS. This study was conducted at a newly constructed WCS under Interstate-90 near Snoqualmie Pass, Washington, in 2017. Our objective was to compare the small mammal species composition at the WCS and adjacent restoration sites to reference sites in the nearby forest. We also sought to evaluate how small mammals used installed habitat features (rock piles, brush piles, or fallen logs) to move through the WCS. We used live-trapping and remote camera-traps to assess small mammal communities. Our results indicated significantly fewer species, lower species diversity, and greater abundance of generalist species in and near the crossing structure than in the reference sites. Small mammals showed no preference for any particular habitat features across all sites but were more likely to be captured near a feature than in open areas. Two years post-construction, the WCS harbored half of the species found in the forest reference sites. We expect the number of small mammal species using the WCS to increase in the future as the habitat develops to support richer biodiversity and as additional species encounter the WCS.**

Key Words.—community ecology; forest specialist; habitat fragmentation; habitat generalist; landscape connectivity; microhabitat; *Peromyscus*; road ecology

INTRODUCTION

Habitat fragmentation, urban sprawl, and expanding transportation networks have fractured ecosystems, restricting the movement of organisms across the landscape (Saunders et al. 1991; Forman and Alexander 1998; Goosem 2000; Dickson et al. 2005; Benítez-López et al. 2010; Buchmann et al. 2013). Traffic noise and lights cause many species to avoid crossing busy highways and connectivity is further reduced by mortality due to wildlife-vehicle collisions (Clevenger et al. 2003; Dodd et al. 2004; Glista et al. 2009; González-Galina et al. 2013). Transportation departments have recognized this rise in human-wildlife conflict and have responded by improving the safety of roads and prioritizing long-term monitoring (Clevenger and Waltho 2000; Clevenger 2012; Andrews et al. 2015). Reduced wildlife connectivity caused by roads at local and landscape scales can be mitigated by a variety of techniques, ranging from posting caution signs in high-risk wildlife crossing zones to constructing wildlife crossing structures (WCS; Hardy et al. 2003; Ford et al. 2009; Beckmann et al. 2012; Andrews et al. 2015; Huijser et al. 2016).

These methods for improving road safety for both drivers and wildlife can often be integrated into existing road construction projects (Clevenger et al. 2001a; Gurrutxaga and Saura 2014; Sawyer et al. 2016). For example, Interstate-90 (I-90) in Washington State bisects the Cascade Range and critical wildlife habitat corridors (Interstate 90 Snoqualmie Pass East Mitigation Development Team 2006). Near Snoqualmie Pass (elevation 921 m), the interstate experiences traffic volumes averaging 31,000 vehicles on a typical day and > 58,000 vehicles on a busy weekend day (Washington State Department of Transportation [WSDOT] (US). 2016. 2016 Annual Traffic Report. Olympia (WA). Washington State Department of Transportation (US). Available from http://www.wsdot.wa.gov/mapsdata/travel/pdf/Annual_Traffic_Report_2016.pdf [Accessed 26 April 2018]). What began as a project to increase the number of lanes and make road safety improvements to accommodate increasing traffic volumes transformed into a collaborative conservation partnership with the added goal of improving the permeability of the interstate to wildlife (Interstate 90 Snoqualmie Pass East Mitigation Development Team 2006). As a result, Washington State Department of Transportation included 27 medium (2 m wide) to large (> 100 m wide) WCS (Mark Norman, pers. comm.) within the I-90 Snoqualmie Pass East Project (Wildlife Working Group 2008). About half of these have been constructed since 2008, with the remainder in planning stages. These WCS are designed to be effective not only for large mammals, but also for small mammals, fish, amphibians, and reptiles.

Small animals, with typically small home ranges and limited dispersal distances, are especially vulnerable to vehicle mortality and reductions in their ease of movement from habitat fragmentation and roads (Bowman et al. 2002; Jenkins et al. 2007; Barthelmess and Brooks 2010; Downs and Horner 2012; González-Gallina et al. 2013). Most wildlife crossing structures have been designed and tested for large-bodied, high-mobility species such as ungulates and large carnivores (Gloyne and Clevenger 2001; Clevenger and Waltho 2005), while studies have only recently begun evaluating the effectiveness of wildlife crossing structures for small mammals (McDonald and St. Clair 2004; D'Amico et al. 2015; Martinig and Bélanger-Smith 2016) and other low-mobility species (Rytwinski and Fahrig 2011; Cunnington et al. 2014). Small mammals (< 5 kg; Bourlière 1975; Merritt 2010) serve important roles as both prey species and predators (e.g., consumers of plants and insects) and provide ecosystem services, such as seed and spore dispersal (Maser et al. 1978; Hayward and Phillipson 1979; Martin 2003; Pearce and Venier 2005). For example, in Pacific Northwest forests, Bushy-tailed Woodrats (*Neotoma cinerea*) and Northern Flying Squirrels (*Glaucomys sabrinus*) are two of the predominant prey species for the endangered Northern Spotted Owl (*Strix occidentalis caurina*; Bevis et al. 1997; Forsman et al. 2001; Cutler and Hays 2016). Some small mammals, such as Northern Flying Squirrels, serve as important dispersal agents for mycorrhizal fungi; loss of these dispersers can negatively affect the long-term function of coniferous forests, especially during recovery after large-scale wildfire or silviculture disturbance (Pyare and Longland 2001; Lehmkuhl et al. 2006).

Our objective was to evaluate small mammal use of one of the first completed WCS of the I-90 Snoqualmie Pass East Project. We aimed to determine which species of small mammals (target species < 1 kg) were using the WCS in its early post-construction stages and how they were moving through the structure. We compared the richness, diversity, relative abundance, community composition, and movement of small mammal species at the WCS to adjacent restoration sites and reference sites in nearby forest. We also evaluated the effectiveness of habitat features (fallen logs, brush piles, rock piles, and snags placed in the WCS and restoration sites) in providing cover for small mammals by testing the likelihood of capturing an animal at different habitat features. We hypothesized that in the early years after construction, WCS harbor only a subset of the surrounding small mammal assemblage that is able to inhabit recently disturbed habitats (Taylor 1999; Smith and Fox 2017), and habitat generalist species are more likely than forest specialist species to be the first to use a WCS. We predicted that the WCS would have lower species richness and lower diversity than the forest reference sites. The restoration sites would have intermediate species richness and diversity compared to the WCS and reference sites because they are closer to the forest edge and have more vegetation than the WCS (Monamy and Fox 2000; Clevenger et al. 2001b; McDonald and St. Clair 2004). We further hypothesized that small mammals are more likely to move along habitat features than in the open (Brehme et al. 2013; Ascensão et al. 2016). We therefore predicted that small mammals would be trapped more frequently in or near habitat features than in more open areas at our study sites. The main outcomes of our study are to provide early post-construction data for future assessments and to suggest specialized habitat restoration improvements for small mammals at future WCS in the I-90 Snoqualmie Pass East Project and other connectivity projects.

Methods

Study area.—We conducted our study along the I-90 transportation corridor in the Cascade Range of Washington State, approximately 4 km east of the Snoqualmie Pass summit near the lower Gold Creek Valley and upper end of Keechelus Lake. The mixed coniferous, mid-elevation forests were dominated by Western Hemlock (*Tsuga heterophylla*), Douglas-fir (*Pseudotsuga menziesii*), and Western Red Cedar (*Thuja plicata*), with Vine Maple (*Acer circinatum*) and alder (*Alnus* spp.) in the forest understory. Riparian areas along streams and lakeshore habitats had interspersed patches of willow (*Salix* spp.) and cottonwood (*Populus* spp.) trees. The patchwork nature of land ownership around the Okanagan-Wenatchee National Forest, Interstate-90, railroad beds, Forest Service roads, and regulated reservoir levels were sources of considerable anthropogenic disturbance.

We compared the small mammal communities at five study sites: a WCS site, two restoration sites, and two forest reference sites (Fig. 1; Table 1). Our WCS was the

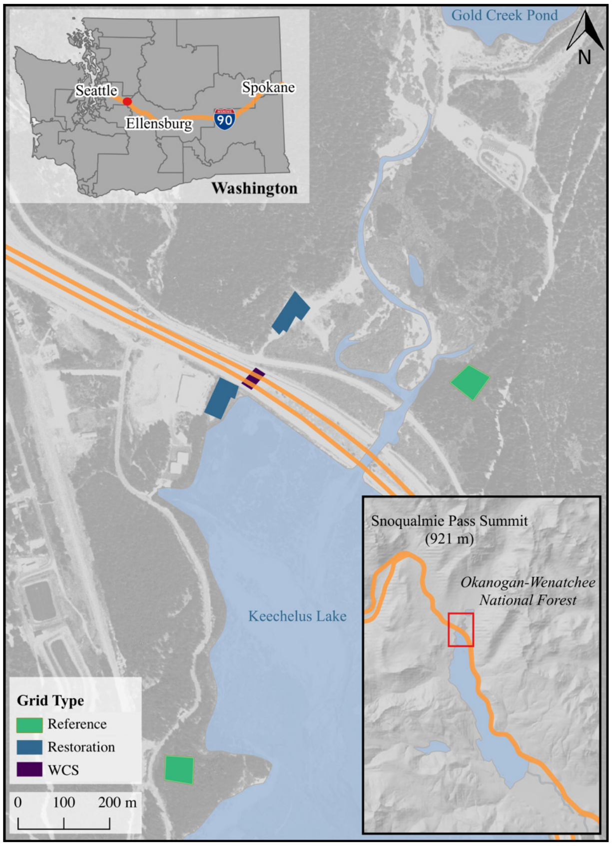

Figure 1. Northeastern portion of the Snoqualmie Pass East Project area and study area (black box in bottom inset) located along Interstate-90 in the Cascade Range of Central Washington State, USA, in summer 2017. The five study sites occurred north and south of I-90 and included the Hyak wildlife crossing structure (WCS).

Table 1. Location and description of study sites in the Cascade Range of Washington, USA, summer 2017: forest reference sites north and south of I-90, restoration sites north and south of I-90, and a wildlife crossing structure (WCS) under I-90. Habitat features included brush piles (BP), fallen logs (FL), rock piles (RP), and snags (S).

					Habitat Features Present			
Site	Latitude	Longitude	Elev. (m)	Grid Area (ha)	BP	FL	RP	S
Reference North	47.3910	-121.3797	772	0.36	x	x		x
Reference South	47.3844	-121.3892	777	0.36	x	x		x
Restoration North	47.3934	-121.3853	763	0.34	x	x	x	x
Restoration South	47.3915	-121.3874	771	0.35	x	x	x	x
WCS	47.3917	-121.3857	768	0.18		x	x	

Hyak wildlife undercrossing, constructed in 2013. The WCS with its 36-m span, soil floor, and solid concrete walls and ceiling allows wildlife to pass underneath I-90 (Fig. 2). In 2015, contractors added bark mulch over the soil, planted sparse ferns and herbaceous plants, scattered large logs throughout, and piled rocks along the interior walls (Fig. 2). At the time of the study, no fencing existed near the structure to guide animals to it or prevent them from attempting to cross the interstate. The two adjacent restoration sites had been previously cleared but were replanted with native trees and shrubs in 2015. Rock piles, snags, fallen logs, and brush piles were added to both sites as habitat features for wildlife. A paved Forest Service road separated the northern restoration site from the WCS. This site was a temporary wetland in early spring. The southern restoration site was directly adjacent to the southern entrance of the undercrossing. The two forest reference sites served as a baseline for expected small mammal species in the area. The northern reference was 200 m northeast of I-90 and southern reference was 800 m west of I-90 (Fig. 1). These second-growth forests had large-diameter conifers, well-developed canopies, and extensive woody debris on the forest floor and experienced occasional human recreation activity.

Live trapping.—At each site we installed a live-trapping grid. The general layout consisted of a 60 × 60 m (0.36 ha) grid with 49 stations in a 7 × 7 array with 10-m spacing. We placed one Sherman live trap (Model LFATDG, H.B. Sherman Traps, Tallahassee, Florida, USA) at every station (49 traps per grid) and one Tomahawk live trap (Model 201, Tomahawk Live Trap, Hazelhurst, Wisconsin, USA) on alternate lines at 20-m intervals (16 traps). Two pitfall arrays, one upland and one lowland, each had one central 3.8 L (1-gallon) bucket and three peripheral buckets at the ends of 5-m galvanized wire mesh drift fence radiating from the center. We made slight modifications to the grid layout at the restoration sites due to spatial constraints, but the total number of traps was maintained. The WCS could accommodate only a half-size grid (30 × 60 m) with 28 Sherman traps, eight Tomahawks, and one central pitfall array.

We visually assessed the type of habitat feature that occurred within a 1-m radius of each grid station and categorized it as rock pile, log, brush pile, snag, or open. Rock piles were several layers of rocks at least 3 m high and at least 3 m in diameter. Fallen logs were at least 0.5 m in diameter, at least 1 m long, with a minimal degree of decomposition. Brush piles were at least 3 m in diameter and multilayered, with several branches of different sizes. Snags were standing dead trees or upright placed logs. We categorized as open any grid stations with habitat features smaller than the specified dimensions or without any habitat features.

We trapped each site for two consecutive nights in mid-July and again in late August 2017, for a total of 1,328

Figure 2. The Hyak wildlife undercrossing located at the Snoqualmie Pass in Washington State is an entirely terrestrial crossing structure with native plants, a wood-mulch floor, different sized fallen logs distributed throughout, and rock piles lining the western and eastern borders. (Photographed by Lindsay Millward).

trap-nights (number of traps set multiplied by number of nights traps were open). Bait for Sherman traps was a mixture of peanut butter, rolled oats, and molasses; for Tomahawk traps we used this mixture plus a piece of carrot. We added 8–10 live mealworms, a bottle cap filled with water, and a toilet paper roll (shelter) to each pitfall bucket to decrease shrew (*Sorex* spp.) mortality (Shonfield et. al. 2013). We set traps in the late afternoon and checked them just before sunset and again in early morning to increase the likelihood of capturing diurnal, crepuscular, and nocturnal species. To avoid overheating animals, we closed all traps during the day.

We brought all captured animals in traps to a shaded processing area. For each individual, we identified the species, took standard measurements (weight, body length, tail length, hind foot length, ear length), recorded sex and age (juvenile, subadult, adult), and applied nail polish to the toes of one foot as a temporary mark. We used tail length to differentiate between adult Keen's Deer Mice, *Peromyscus keeni* (tail vertebrae length $\geq$ 96 mm), and Deer Mice, *P. maniculatus* (tail vertebrae length < 96 mm; Gunn and Greenbaum 1986; Zheng et al. 2003). We differentiated juveniles, subadults, and adult mice by weight and pelage color: juveniles weigh 13–16 g and have a solid grey pelage and adults weigh > 17 g and have a brown, course pelage (Healey 1967; Sullivan 1979; Van Horne 1982). Differentiating subadult deer mice is challenging and we made our best judgement calls in the field based on size and pelage (color and coarseness). We used an N165 injector needle (16 GA) to inject an 8 mm PIT tag (Biomark mini-HPT8, Boise, Idaho, USA) subdermally behind the head, roughly between the scapulae. We then released animals at the site of capture. Any individuals that died in the traps or during processing, we collected as specimens and stored in a freezer.

***Remote camera traps.*—**We used remote camera traps (models RapidFire, Silent Image, Hyperfire, and Convert IR; Reconynx, Holmen, Wisconsin, USA) for a two-week period between July and October 2017 at each site to capture species that might not readily enter live traps. To avoid disturbance from researcher presence, we installed cameras at sites when we were not actively trapping. On each grid we placed four cameras per habitat feature (fallen log, brush pile, rock pile, and snag) at positions deemed most suitable for both camera position and concealment from potential theft. We mounted cameras within habitat features or facing them (depending on the feature) and did not use a scent lure (Glen et al. 2013). We set cameras to capture a burst of three or five images within a 5-sec interval when triggered by an animal, with a 30-sec delay between bursts (De Bondi et. al. 2010). We counted images of the same species more than one hour apart as separate visitation events (Derugin et. al. 2016). In cases where we could not identify the images to species, we placed it into a species group designation (e.g., woodrats, deer mice).

***Data summary and statistical analysis.*—**We classified species recorded during the study as either habitat generalists or forest specialists (Appendix 1; Naughton 2012). We constructed species accumulation curves (Ugland et al. 2003) to verify sufficient sampling of the small mammal assemblage before further analysis. For each site type (WCS, restoration grids, reference grids) we calculated Shannon diversity index (H')

$$H' = \Sigma\, p_i \ln p_i$$

where p_i is the proportion of individuals found of species i, and Shannon's equitability (a measure of evenness; E_H)

$$E_H = H' / \ln S$$

where S is the number of species sampled (Magurran 1988; Heip et al. 1998). We determined species richness, or number of species sampled, for each site type by combining data from live-trapping and cameras. We used a Hutcheson *t*-test to compare species diversity between the restoration and reference sites (Hutcheson 1970). Because the sampling effort for the WCS (i.e., only one replicate) was not equivalent to the other sites, we did not include it in the *t*-test. For other comparisons, we standardized capture data (to account for fewer trapping stations in the WCS) as number of captures per 100 trap-nights (number of captures or camera-trap visitations divided by the number of trap-nights, multiplied by 100) or by comparing relative abundance (number of individuals divided by the total number of captures at each site; Mengak and Guynn 1987; dos Santos-Filho et al. 2006). We combined camera and live-trap data only to determine species richness at each site; otherwise, the two capture methods were analyzed separately.

We conducted all other statistical analysis using R 3.5.1 (R Core Team 2018). To evaluate habitat preference by small mammals, we used a Generalized Linear Mixed Model (GLMM) implemented with the glmmTMB package in R (v0.2.3; Brooks et al. 2017). The total number of individuals per trap station was modeled as a function of habitat feature (fallen log, brush pile, rock pile, open), trap type (Sherman, Tomahawk, pitfall), site type, and individual site, with site treated as a random effect (Hamilton et al. 2015). We used a log-link function and a Conway-Maxwell-Poisson distribution of error terms, due to significant underdispersion. We classified site type as either reference or combined restoration/WCS because the wildlife crossing structure was represented by only one site. Post-hoc, we replaced habitat feature with presence/absence of any habitat feature to produce our final model. We performed model validation via simulation using the DHARMa package in R (v0.2.0; Hartig 2018). Validation of our final model revealed no evidence of model misspecification (one-sample Kolmogorov-Smirnov test on the residuals, $D = 0.05$, P

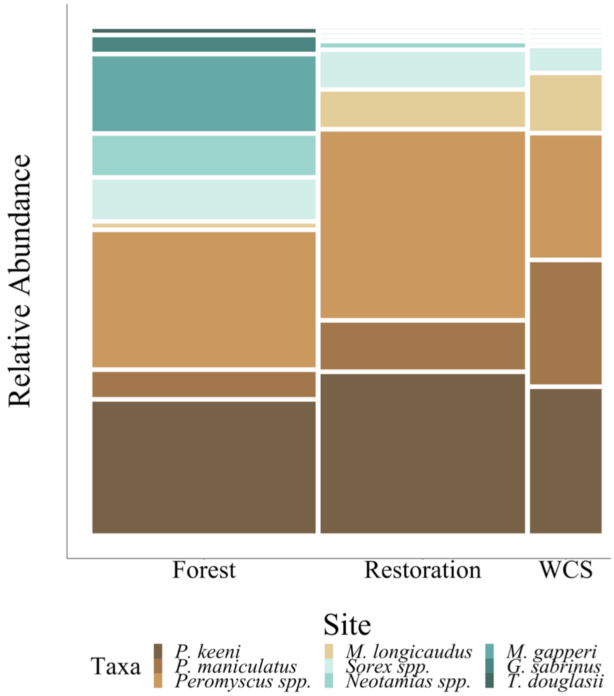

Figure 3. Mosaic plot of small mammal community composition at forest, restoration, and wildlife crossing structure sites in summer 2017 near the Snoqualmie Pass, Washington State. Relative abundance is the number of individuals of a particular species divided by the total number of individuals at each site. Column width represents the sampling effort. Generalist species are represented by shades of brown, while forest specialist species (Naughton 2012) are represented by shades of blue. (Appendix 1).

= 0.340). Following best practices outlined in Bolker et al. (2009), we estimated parameters by maximum likelihood laplace approximation and performed hypothesis testing of fixed effects with Wald *Z* tests. The overall significance of factors with more than two levels was assessed using a Likelihood Ratio test, followed by specific post-hoc pairwise comparisons via Wald *Z* tests using the glht function from the multcomp package in R (v1.4-10; Hothorn 2008) and the Tukey method to adjust for multiple testing.

To evaluate distance traveled by small mammals within a site, we calculated straight-line (i.e., minimum) distances between trapping stations for each individual that we recaptured on consecutive nights. We applied a General Linear Model executed with the lm function from the stats package in R 3.5.1 (R Core Team 2018) to model distance traveled in one night as a function of site, species, sex, and age. We used a power transformation of 0.7 on distance traveled to yield maximum conformation to a Gaussian distribution of error terms, as confirmed by model validation via simulation using the DHARMa package in R (v0.2.0; Harting 2018; one-sample Kolmogorov-Smirnov test on the residuals, $D = 0.09$, $P = 0.540$). We used *F*-tests for hypothesis testing of fixed effects.

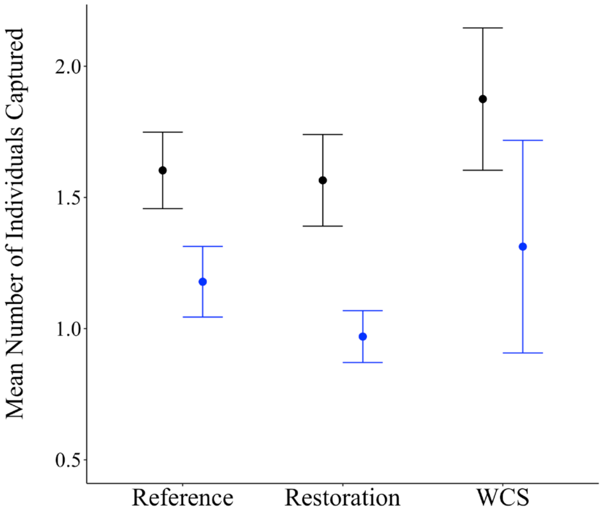

Figure 4. Mean (± standard error) number of small mammal individuals captured per 100 trap-nights at trap stations with habitat features present (black) and absent (blue) during 2017 at all sites near the Snoqualmie Pass, Washington State.

RESULTS

Live trapping.—Among the 472 total captures, we recorded 293 unique individuals and 10 species. Deer Mice and Keen's Deer Mice were the most frequently captured species, composing 69% of captures at the reference sites, 83% at the restoration sites, and 86% at the WCS (Figs. 3 and 4). The relative abundance of generalist species increased from the reference sites to restoration sites to the WCS, while the relative abundance of specialist species decreased (Table 2; Appendix 1). Compared to the reference sites, the WCS had substantially higher relative abundances of generalist Deer Mice (> 5 times as high; 6.9 vs. 1.2 individuals per 100 trap-nights), Keen's Deer Mice (1.5 times as high; 9.4 vs. 6.3 individuals per 100 trap-nights), and Long-tailed Voles, *Microtus longicaudus* (15 times as high; 3.1 vs. 0.2 individuals per 100 trap-nights). Within the WCS, several individuals were recaptured on different nights (one of five Long-tailed Voles, 17 of 39 deer mice). Two (of three) female Long-tailed Voles and three (of 18) female Deer Mice were reproductively active (pregnant, nipples prominent, or lactating).

Small mammal community composition.—Species accumulation curves leveled off, indicating sufficient sampling (Appendix 2). Species richness varied across the three site types (Table 3). Three species (Keen's Deer Mouse, Deer Mouse, and Montane Shrew, *Sorex monticolus*) occurred in all site types, and five other species (Cinereus Shrew, *Sorex cinereus*; Long-tailed Vole; Townsend's Chipmunk, *Neotamias townsendii*; Douglas Squirrel, *Tamiascurius douglasii*; and Snowshoe Hare, *Lepus americanus*) occurred at both restoration and reference sites, but not at the WCS (Fig. 3). We captured

Table 2. Total number of trap-nights, recapture rates, and total number of captures per 100 trap-nights including number of individuals in parentheses at each site from live-trapping in 2017 near Snoqualmie Pass in the central Cascade Range of the central Cascade Range in Washington State. Habitat specialization (generalists and forest specialists) from Naughton (2012; Appendix 1).

Site Type	Trap Nights	Generalists	Forest Specialists	Total Captures	Recapture Rate
Reference	584	25.2 (11)	10.4 (2)	35.6 (13.0)	36%
Restoration	584	28.8 (9.9)	2 (0.4)	30.8 (10.3)	33%
WCS	160	43.1 (16.9)	1.3 (0)	44.4 (16.9)	38%

Table 3. Small mammal species richness and evenness (Shannon's equitability, E_H) from live trapping at sites in 2017 near Snoqualmie Pass in the central Cascade Range in Washington State. Combined reference and combined restoration are the combined richness and evenness of the north and south sites.

Site	Richness	E_H
Combined Reference	11	0.74
Reference North	9	0.63
Reference South	8	0.88
Combined Restoration	8	0.62
Restoration North	7	0.40
Restoration South	4	0.69
WCS	4	0.60

Southern Red-backed Voles (*Myodes gapperi*), Yellow-pined Chipmunks (*Neotamias amoenus*), and Northern Flying Squirrels (*Glaucomys sabrinus*) exclusively at the reference sites.

The reference sites had a significantly higher diversity than the restoration sites ($t = 4.382$, df = 140, $P < 0.001$). The abundances of species in the reference forest community were also distributed more evenly than in the restoration sites (Table 3; Fig. 3). The southern reference site had the highest evenness and highest diversity. The northern restoration site had the least even community, mostly dominated by Keen's Deer Mice, and the lowest diversity index.

Remote camera traps.—We documented 351 small mammal visitations (photo-captures of a species at least 1 h apart) from a total of 838 animal photographs over 761 trap-nights. Across all sites, 42% of visitations were deer mice, 36% were voles (*Microtus longicaudus* or *Myodes gapperi*), 8% were chipmunks, 5% were Douglas Squirrels, and < 10% were shrews, Snowshoe Hares, and Northern Flying Squirrels. We captured seven genera at the reference sites, six genera in the restoration sites, and three genera in the WCS. We captured Snowshoe Hares exclusively on wildlife camera, contributing to the overall species richness documented in the reference and restoration sites.

The total number of photo-captures per 100 trap-nights increased from the reference to restoration sites and WCS (Table 4). At both the WCS and restoration sites, the cameras captured more small mammals at rock piles than at other features. Across all sites, the cameras captured fewer individuals at snags than at other features.

Habitat features.—Based on live-trapping, small mammals collectively showed no significant preference among habitat features (fallen logs, brush piles, rock piles, and open areas) across all sites (GLMM estimates assessed via likelihood ratio test, LR = 5.183, df = 3, $P = 0.160$). Captures of small mammals, however, were more likely at stations with a habitat feature present than at stations in open areas (Table 5; Fig. 4). Trap type was significant in both models because we captured significantly more animals in Sherman traps than either Tomahawk traps or pitfall traps (Table 5). Overall, capture rates did not differ between reference sites and restoration/WCS sites (Table 5).

Intra- and inter-grid movement.—The disproportionate composition of species with movement events (76 out of 84 events are deer mice) resulted in species being a significant predictor in our model ($F_{6,77} = 2.431$, $P = 0.034$). Therefore, we cannot confidently describe movement rates or trends between species. Of all species combined, neither site type ($F_{4,79} = 1.444$, $P = 0.230$) nor sex ($F_{1,82} = 0.208$ $P = 0.645$) was a significant predictor of distance traveled in one night within grids, but age was a significant factor ($F_{1,82} = 5.884$, $P = 0.018$), with adults of all species traveling farther than subadults/juveniles (estimated effect size of adult age = 6.2 m, 95% confidence interval = 0.6–14.6 m). Of the 84 total movement events that fit our criteria, 50% were adults and 52% were males. Small mammals moved on average 16.8 m (± 2.4) per night in the reference grids, 17.1 m (± 2.6) in the restoration grids, and 15.5 m (± 3.0) in the WCS site.

Movements of animals between sites were documented six times; all were deer mice. Five mice moved an average distance of 39 m from the WCS to the southern

Table 4. Average number of days a camera was deployed at a site, number of total camera-traps at a site, total number of trap-nights, and total number of captures per 100 trap-nights for each habitat feature category in 2017 near Snoqualmie Pass in the central Cascade Range of Washington State.

				Total Captures per 100 trap-nights				
Site	Average Days	No. Camera Stations	Trap Nights	Rock Pile	Brush Pile	Fallen Log	Snag	All Features
Reference	16	21	339	—	38.5	37.5	3.5	26.5
Restoration	14	21	294	75.9	53.6	42.9	14.3	54.8
WCS	16	8	128	115.6	—	28.1	—	71.9

Table 5. Fixed effect size estimates, 95% confidence intervals (C.I.), and *Z* and *P* values from a Generalized Linear Mixed Model where total number of small mammals captured in 2017 at each trap throughout the study period was modeled as a function of presence/absence of habitat features, trap type (Sherman, Tomahawk, pitfall), site type, and individual site, with site treated as a random effect. Effect sizes and confidence intervals are reported on the log scale. Asterisks (*) indicate significant effects, as assessed by Wald *Z* tests, using a Tukey correction for multiple testing. Comparisons are HF-NHF =habitat feature, no habitat feature, ST-TT = Sherman versus Tomahawk traps, ST-PT = Sherman versus pit-fall traps, TT-PT = Tomahawk versus pit-fall traps, and R-R&WCS = reference site versus restoration and wildlife crossing structure sites.

Comparison	Estimated Effect Size	95% C.I.	*Z*	*P*
HF-NHF	0.17	± 0.16	2.13	0.033*
ST-TT	2.01	± 0.44	8.88	< 0.001*
ST-PT	2.74	± 0.96	5.58	< 0.001*
TT-PT	0.73	± 1.05	1.36	0.342
R-R&WCS	0.05	± 0.17	0.63	0.530

restoration site. One female mouse traveled 136 m from the northern restoration site, across a Forest Service road, to the WCS.

DISCUSSION

The main objective of this study was to compare small-mammal species richness, composition, relative abundance, and movement at a WCS to adjacent restoration sites and nearby forest reference sites. Our data supported the prediction that the WCS would have lower species richness than both the restoration and reference sites, with the restoration sites having intermediate species richness. In addition to being exposed to noise, lights, and pollution caused by the highway, the WCS was < 2 y old at the time of this study and was in the early stages of habitat recovery with underdeveloped vegetation. This lack of habitat complexity likely contributed to the lower species richness and diversity we observed. For example, chipmunks prefer a thicker understory because it provides a source of food and cover from predators whereas Northern Flying Squirrels prefer a closed canopy but open forest floor that provides greater ease of movement (Carey 2000; Zwolak 2009; Rotholz and Mandelik 2013).

Several of the small mammal species recorded in our reference sites, but not in the WCS, such as the Yellow-pine and Townsend's Chipmunks, Douglas Squirrels, and Snowshoe Hares are forest habitat specialists. Habitat specialists can be key indicators for environmental changes because they are especially sensitive to any changes to their habitats, unlike their generalist counterparts that can use a larger variety of resources (Henrik 1994). Specialists may be more vulnerable than generalists to the effects of habitat fragmentation and road-effect zones because they are less tolerant of environmental changes and are less likely to occupy the degraded habitat often found adjacent to roads (Umetsu and Pardini 2006; Barthelmess and Brooks 2010; Rotholz and Mandelik 2013). Most forest specialist small mammals require shrub or canopy cover and coarse woody debris for habitat, foraging, and safer movement (Carey and Harrington 1995; Gitzen and West 2001; Sollmann et al. 2015). Therefore, to be effective for the broadest spectrum of small mammal species, WCS that are large enough to accommodate habitat features should contain ample cover from predators, such as fallen logs and understory vegetation, to facilitate movement and improve connectivity (Kintsch and Cramer 2011; D'Amico et al. 2015; Schlinkert et al. 2016).

The species composition of small mammals varied among the site types we studied. Habitat generalists were the main inhabitants of the WCS. Similarly, only *Peromyscus* spp., among several small mammal species, were recorded passing through two WCS in Vermont 2–3 y post-construction (Bellis et al 2013). Habitat degradation, such as forest clear-cutting and agriculture, alters the composition of small mammal communities, favoring open-habitat species such as Deer Mice and Long-tailed Voles (Zwolak 2009; Panzacchi et al. 2010). These species can thrive in early-successional, high-disturbance environments (Manson et al. 1999; Goosem 2000; Bissonette and Rosa 2009) like the recently restored areas and WCS in our study. Our forest reference sites, in contrast, supported a more even, diverse assemblage of species mostly consisting of closed-canopy, or mature forest, specialists.

The presence of high-quality habitat, such as natural substrate, natural light, habitat features, and plentiful vegetation, can encourage WCS use by small mammals (Ford and Fahrig 2008; McGregor et al. 2008; Hennessy et al. 2018). Our results showed an increased likelihood of captures when traps were located in or near any habitat feature (fallen logs, brush piles, rock piles) rather than in open or featureless areas. This apparent lack of preference for specific features may be linked to the high abundance of generalist species found in our surveys because Deer Mice and Long-tailed Voles are able to thrive in a broad variety of environmental conditions and habitat types. Our finding of more captures near habitat features aligns with other studies showing that Deer Mice are more likely to travel along logs and branches than on open ground (Graves 1988; Carey and Harrington 2001). Predators of small mammals, such as coyotes, have begun using the undercrossing (Josh Zylstra, pers. comm.), so movement across open areas may be risky and the protection offered by any habitat feature may be sufficient, at least for generalist species (Tallmon et al. 2003). As these restoration sites mature and the density of vegetation increases, we may begin to see species-specific habitat preferences and increased use by forest specialists (Smith et al. 2015). Captures of two forest specialists at our sites in 2018 (Southern Red-backed Vole in the WCS; Pacific Jumping Mouse, *Zapus trinotatus*, at the northern restoration site) support this prediction.

A major drawback to our study design was that we evaluated the use of only one WCS; however, sampling effort within the WCS appeared sufficient (species accumulation curve approached an asymptote; Ugland et al. 2003). Additionally, the high abundance of Deer Mice caught in Sherman live traps may have biased the survey by physically limiting other species from being caught in the same traps. Remote cameras showed a lower relative abundance of Deer Mice, indicating the assemblage may have a more even composition than estimated by live trapping alone. The fewer camera trap-nights, camera-trap malfunction or poor placement, and inability to identify individuals prevented robust statistical analysis of our camera-trap data. The patterns observed from our remote camera-trap data provide an excellent opportunity for future studies. Movement patterns of small mammals near roads and within crossing structures, and health of individuals as an indicator of fitness at these sites of higher disturbance also merit further exploration (Sollman et al. 2015; Grilo et al. 2018).

This study is the first evaluation of wildlife use of crossing structures in the I-90 Snoqualmie Pass East Project. We expect that small-mammal richness and diversity will increase over the next several years as more species from the nearby forest discover and explore the WCS. Wildlife may require several years to adapt to WCS (Clevenger and Waltho 2005; Ford et al. 2009), and the number of animals discovering and using WCS often increases with time (Bond and Jones 2008; Gagnon et al. 2011; Soanes et al. 2013; van der Grift and van der Ree 2015). Development of the planted vegetation over time should also promote higher species richness and diversity in and near the WCS due to increased cover (McDonald and St. Clair 2004), food resources, and habitat complexity (MacArthur and MacArthur 1961; Bell et al. 1991).

Unlike large mammals with larger home ranges, such as Elk (*Cervus canadensis*) in the Cascade Range with 1,300–11,000 ha home range size (McCorquodale 2003), that likely use WCS primarily to move across the landscape, small mammals, which have smaller home range sizes (e.g., 0.1–0.4 ha for shrews, Hawes 1977; 0.4–4.0 ha for deer mice, Stickel 1968), may also be living largely within and adjacent to these newly created habitats. Our recaptures of individuals and captures of reproductive individuals within and adjacent to the WCS suggest that these small mammals are living in, rather than just passing through, the WCS. The availability of WCS is critical for improving connectivity between habitat patches separated by highways (Ford and Clevenger 2018). Therefore, designing WCS that contain high-quality, suitable habitat may be important to ensure these connectivity mitigation structures are effective at improving the permeability of the interstate to a variety of small mammal species that reside in these forested mountain habitats.

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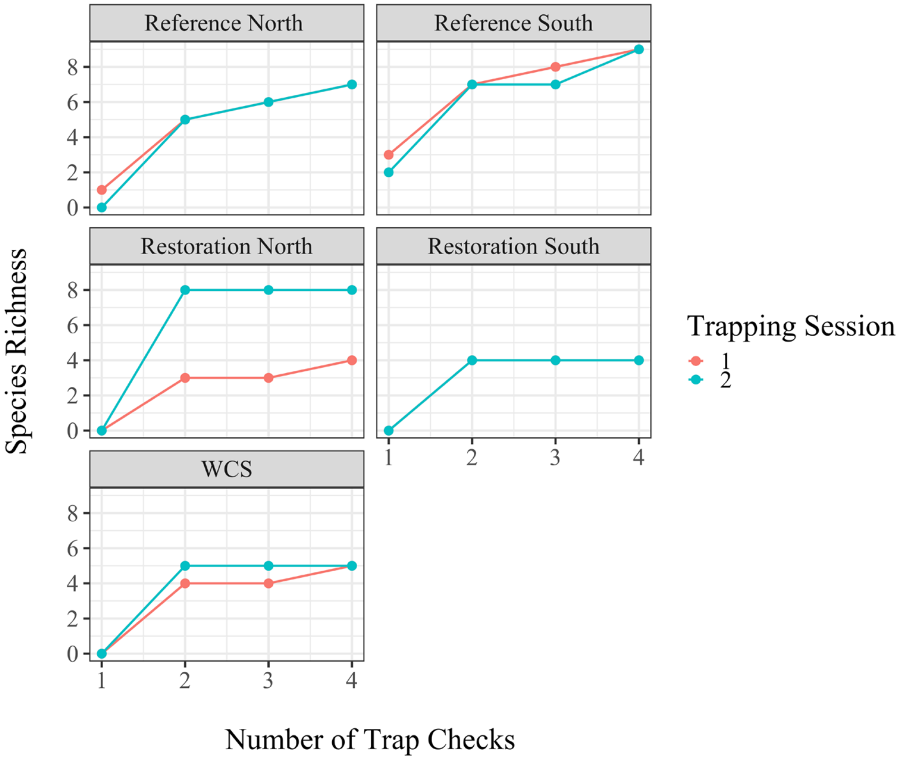
Appendix 1. Scientific and common names of small mammal species recorded during the study, and their designation as habitat generalists (G) or forest specialists (S; from Naughton 2012).

Scientific Name	Common Name	Habitat Generalist or Forest Specialist
Sorex cinereus	Cinereus Shrew	*S*
Sorex monticolus	Montane Shrew	*G*
Sorex sp.	unidentified Shrew	
Peromyscus keeni	Keens's (Pacific) Deer Mouse	*G*
Peromyscus maniculatus	Deer Mouse	*G*
Microtus longicaudus	Long-tailed Vole	*G*
Myodes gapperi	Red-backed Vole	*S*
Neotamias amoenus	Yellow-pine Chipmunk	*S*
Neotamias townsendii	Townsend's Chipmunk	*S*
Glaucomys sabrinus	Northern Flying Squirrel	*S*
Tamiasciurus douglasii	Douglas's Squirrel	*S*
Lepus americanus	Snowshoe Hare	*S*



Appendix 2. Species accumulation curves or count of cumulative number of species from the start to the end of trapping, for each site as a reflection of trapping effort. Each site was trapped in two sessions during 2017 at sites near Snoqualmie Pass, Washington: once at the beginning of summer and again at the end of summer.

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NOTES

MYOTIS BAT MORTALITY CAUSED BY THE PLANT FULLER'S TEASEL (*DIPSACUS FULLONUM*)

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***Abstract*.—Knowledge of the risks of invasive plants to bats can inform bat stewardship practices. This note reports the deaths of a Long-eared Bat (*Myotis evotis*) and a California Myotis (*Myotis californicus*) caused by the plant Fuller's Teasel (*Dipsacus fullonum*).**

Key Words.—bats; British Columbia; conservation; invasive plants; teasel

Bat populations in North America have been seriously impacted by White Nose Syndrome (a fungal infection), and bats worldwide face a multitude of other threats, including pesticides, habitat loss, wind farms, heat stress, harassment from humans, and invasive species (Frick et al. 2020). In response, bat conservation programs have encouraged bat-friendly communities (British Columbia Community Bat Program. 2018. Bat-friendly communities: a guide for managing and enhancing bat habitat in British Columbia. British Columbia Community Bat Program, Canada. 60 p. Available from https://www.bcbats.ca/images/BC-Bat-friendly-Communities-Guide-2018.pdf [accessed 10 June 2020]) that promote bat stewardship education and habitat protection strategies that can be applied by everyday citizens. One such strategy is to remove objects near bat roosts that may cause entrapment, such as open barrels of water, open pails, old tires, barbed wire coils, mouse traps located on high ledges, and ornamental and exotic plants that are spiny or adhesive. Examples of plants entrapping bats include Desert Rock Nettle (*Eucnide urens*) trapping a California Leaf-nosed Bat (*Macrotus californicus*; Stager 1943), rose thorns trapping a Brown Big-eared Bat (*Plecotus auritus*; Venables 1944), Spanish Moss (*Tillandsia usneoides*) strangling a Seminole Bat (*Lasiurus seminolus*; Dunaway 1960), cacti impaling the California Myotis (*Myotis californicus*; Gronau 2006) and the Serotine Bat (*Eptesicus serotinus*; Merzlikin 2017), adhesive spines of tick trefoil (*Desmodium* sp.) entangling the Diminutive Serotine Bat (*Eptesicus diminutus*; Jacomassa et al. 2017), thorns of Buckthorn (*Prunus spinosa*) trapping the Noctule (*Nyctalus noctula*), and sticky Horse Chestnut buds (*Aesculus hippocastanum*) snaring the Brown Big-eared Bat (*Plecotus auritus*; Merzlikin 2017). Burdock (*Arctium* spp.) has trapped the Little Brown Bat (*Myotis lucifugus*; Lyon 1925; Verts 1988), Eastern Red Bat (*Lasiurus borealis*; Johnson 1933), Long-eared Bat (*Myotis evotis*; Hendricks et al. 2003), Silver-haired Bat (*Lasionycteris noctivagans*; Norquay et al. 2010), and pipistrelles (*Pipistrellus* sp.; Merzlikin 2017).

I report here a case in which the plant Fuller's Teasel (*Dipsacus fullonum*) caused the death of two bats, a Long-eared Bat and a California Myotis, on Salt Spring Island, British Columbia, Canada. Teasel is an introduced Eurasian biennial flowering plant that inhabits roadsides and meadows in British Columbia (Douglas et al. 1999). The egg-shaped capitulum, or flower head inflorescence, can be several centimeters in length and includes an array of protruding spines formed from involucral and receptacular bracts. The teasel plants were situated within 20 m of a bat roost, and I observed the two dead bats together on teasel capitula in September 2018, with spines penetrating the patagia at multiple sites. The Long-eared Bat may have contacted the teasel while gleaning an insect. Faure and Barclay (1994) observed that this bat species while gleaning did not produce a feeding buzz and often did not employ echolocation, a lack of which would impair the ability of the bat to resolve the teasel spines. The California Myotis is an aerial hawker that flies close to vegetation (Krutzsch 1954) and may have used the teasel as a perch.

The presence of two bats of different species together is not unexpected as netted bats have been observed to emit distress calls that appear to attract other species (Dechmann and Safi 2005). I have observed the presence of teasel near a number of large bat colonies, and, unfortunately, the cultivation of teasel is promoted by conservation organizations as a means of attracting birds. While the overall risk to bats is unknown, it may be prudent to remove teasel from areas of high bat density, such as in the vicinity of large bat roosts.

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TIPTON KANGAROO RATS (*DIPODOMYS NITRATOIDES NITRATOIDES*), SHRUBS, AND INTRAGUILD COMPETITION

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***Abstract*.—The Tipton Kangaroo Rat (*Dipodomys nitratoides nitratoides*: TKR) is an imperiled species in the San Joaquin Valley of California. Habitat relationships and competition with other kangaroo rat species are still being defined. We examined the relationship between shrubs and TKR as well as the more common Heermann's Kangaroo Rats (*D. heermanni*: HKR) on two live-trapping plots at a study site on the Kern National Wildlife Refuge. The mean distance from capture locations to the nearest shrub was similar for both kangaroo rat species although HKR clearly traveled farther from shrubs (maximum distance > 50 m for HKR versus 14.5 m for TKR). The proportion of stations where only TKR were captured (28.8%) was higher than the proportion where both TKR and HKR were captured (11.0%) suggesting avoidance of the larger HKR by TKR. The proportion of stations with captures of TKR, HKR, or both species did not differ based on the particular species of nearest shrub. Thus, neither kangaroo rat species exhibited an association with a particular shrub species, although TKR were more abundant on the plot where Seepweed (*Suaeda nigra*) was more abundant. Seepweed does not appear to influence microhabitat use by TKR, but Seepweed may be an indicator of habitat suitability for TKR. Shrubs may facilitate the occurrence of TKR in areas where HKR, a larger intraguild competitor, also occurs, and this possibility warrants further investigation.**

Key Words.—California; facilitated coexistence; kangaroo rats; Kern National Wildlife Refuge

INTRODUCTION

Tipton Kangaroo Rats (*Dipodomys nitratoides nitratoides*: TKR) are endemic to the southern San Joaquin Valley in central California (U.S. Fish and Wildlife Service [USFWS] 1998). TKR once were widely distributed on the Valley floor from about the Kings River in Kings County in the north down to the southern end of the Valley in Kern County. They occur in arid scrub habitats, but much of this habitat has been converted to agricultural, urban, and industrial uses (USFWS 1998). By 1985, only an estimated 3.7% of historical habitat remained, and many of these lands consisted of small, isolated fragments of varying quality (Williams and Germano 1992). Due to this profound habitat loss, fragmentation, and degradation, TKR were Federally listed as Endangered in 1988 and by California as Endangered in 1989 (USFWS 1998).

According to the species account in the recovery plan for San Joaquin Valley upland species (USFWS 1998), TKR occur in arid-land communities with generally level terrain and soils with higher salinity. Shrubs typically present include Spiny Saltbush (*Atriplex spinifera*), Desert Saltbush (*Atriplex polycarpa*), Arrowscale (*Stutzia covillei* [= *Atriplex phyllostegia*]), Quailbush (*Atriplex lentiformis*), Iodine Bush (*Allenrolfea occidentalis*), Pale-leaf Goldenbush (*Isocoma acradenia*), and Honey Mesquite (*Prosopis glandulosa*). Seepweed (*Suaeda nigra* [= *S. moquinii* and *S. fruticosa*]) is described as a conspicuous semiwoody species in areas with TKR. Shrub cover typically is sparse to moderate in areas with high TKR density. Cypher et al. (2017. Conservation of endangered Tipton kangaroo rats (*Dipodomys nitratoides nitratoides*): status surveys, habitat suitability, and conservation strategies. Endangered Species Recovery Program. Available from http://esrp.csustan.edu/publications/pdf/Cypher_etal_2017_TKR_Conservation.pdf [Accessed 8 April 2020]) found that TKR were particularly associated with alkali sink communities where Iodine Bush and Seepweed were the dominant shrubs. Iodine Bush was present on 40.0% of the sites where TKR were captured while Seepweed was present on 73.3%.

Heermann's Kangaroo Rats (*D. heermanni*: HKR), a non-protected species, are sympatric with TKR. HKR are larger than TKR (56–74 g versus 35–38 g; Jameson and Peeters 1988; USFWS 1998) and potentially competitively exclude TKR (Williams and Germano 1992; USFWS 1998; Tennant and Germano 2013). Also, Nelson et al. (2007) found that HKR were significantly more abundant on traplines in areas with shrubs compared to traplines without shrubs. Conversely, Short-nosed Kangaroo Rats (*D. n. brevinasus*), which are conspecific with and ecologically similar to TKR, were more abundant in areas without shrubs. Thus, HKR may be more closely associated with shrubs than smaller kangaroo rats like TKR. We assessed the association between shrubs and both HKR and TKR

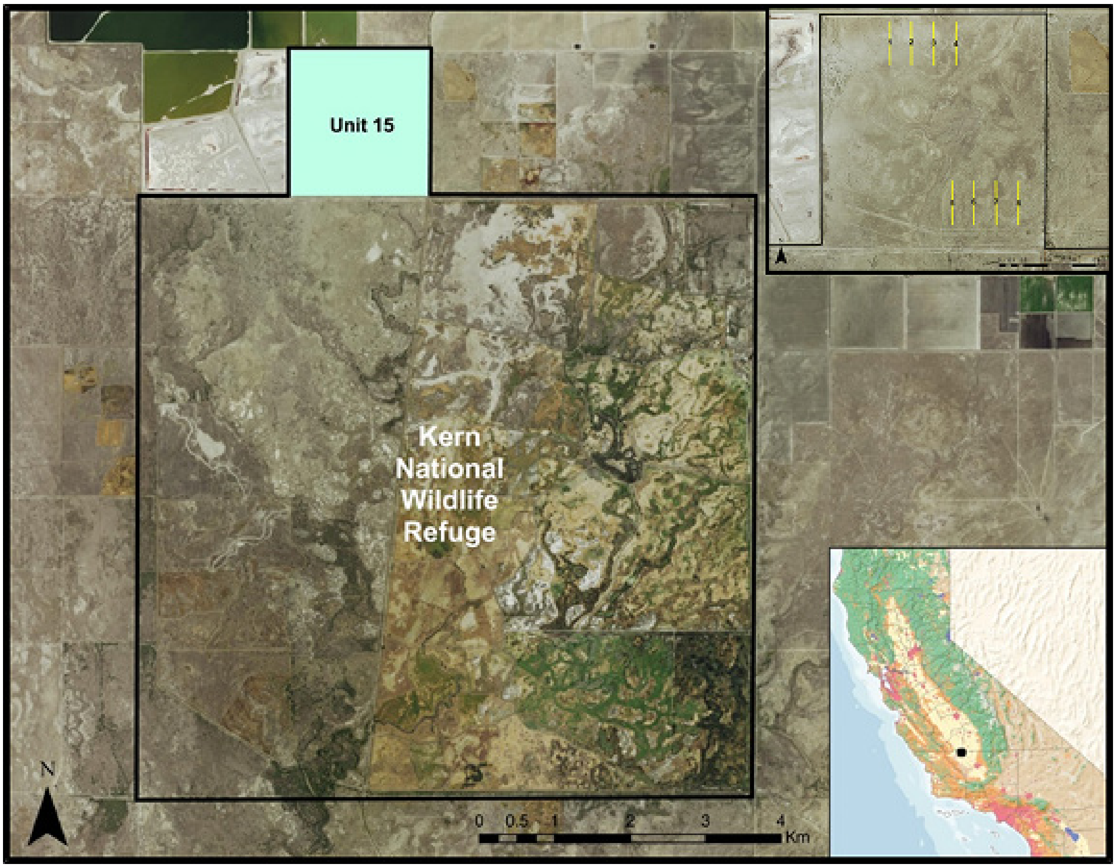

Figure 1. Unit 15 study area at the Kern National Wildlife Refuge in Kern County, California. The top inset shows the location of the two study plots on Unit 15, each consisting of four traplines.

at the scale of specific trap locations on a study site in alkali sink habitat at the Kern National Wildlife Refuge (KNWR). We hypothesize that TKR are not as closely associated with shrubs for cover as are HKR. We predict that the distance between trap locations and shrubs will be greater for TKR than for HKR. We also hypothesize that TKR are associated with alkali sink communities characterized by Iodine Bush and Seepweed shrubs. We predict that TKR will be more frequently captured where Seepweed is abundant.

Methods

Study area.—Our study was conducted on Management Unit 15 at the KNWR, in northern Kern County, California (Fig. 1). KNWR is located approximately 32 km west of the town of Delano. Unit 15 is a parcel approximately 255 ha in size on the north side of the KNWR. KNWR is on the floor of the San Joaquin Valley and the terrain is flat. Mean elevation averages 67 m. The soils in Unit 15 are primarily Twisselman clay and Nahrub clay-lenthent silt loam complex (USFWS 1985, 2005). The regional climate is Mediterranean with hot, dry summers and cool, wet winters. At the town of Wasco, 25 km to the southeast, the average high temperature in August was 37.8° C and the average low was 18.2° C, and the average high temperature in January was 13.8° C and the average low was 1.6° C. Average yearly rainfall was 183.1 mm with most falling from November through April (https://wrcc.dri.edu/cgi-bin/cliMAIN.pl?ca9452).

Vegetation communities in Unit 15 were broadly classified as Valley Sink Scrub (USFWS 2005). The Valley Sink Scrub community was further classified as Valley Iodine Bush Scrub (USFWS 1985). This community consisted primarily of widely spaced shrubs including Iodine Bush, Seepweed, and Alkali Heath (*Frankenia salina*). The California Native Plant Society classifications for this community are the Bush Seepweed Scrub Alliance and the Iodine Bush Scrub Alliance (Sawyer et al. 2009). Common herbaceous plants included brome grasses (*Bromus* spp.), barley (*Hordeum* spp.), annual fescues (*Festuca* spp.), Bird's-eye Gilia (*Gilia tricolor*), Spikeweed (*Centromadia* [= *Hemizonia*] *pungens*), and goldfields (*Lasthenia* spp.). On Unit 15, ground cover typically is sparse (Fig. 2).

Rodents and shrubs.—We live trapped rodents on two plots in Unit 15 (Fig. 1); one plot was in the northern half of the unit and one was in the southern half. On each plot, we established four traplines spaced 150 m apart. Each line was oriented north-south and consisted of 20 medium Sherman box traps (30.5 × 9.5 × 7.6 cm; XLK Extra-Large Kangaroo Rat Trap, H.B. Sherman

Figure 2. Typical shrub density and fall ground cover conditions on the Unit 15 study area, Kern National Wildlife Refuge, California. The shrubs visible in the image are all Iodine Bush (*Allenrolfea occidentalis*). The south plot is shown but conditions on the north plot were similar. (Photographed by Brian Cypher).

Traps Inc., Tallahassee, Florida) spaced 15 m apart. We opened the traps within 2 h of sunset and we provisioned each trap with a handful of millet seed and two sheets of crumpled unbleached paper towel for insulation and preoccupation. We checked and closed the traps the following morning within 2 h of sunrise. For the first capture of each animal, we recorded species, sex, age, and reproductive condition, and we marked each animal ventrally with a non-toxic felt-tipped pen. At first capture, we weighed kangaroo rats and released them at the capture site. We trapped for rodents 30–31 October 2018 for a total of 160 trap-nights on each plot.

We measured the distance (to 0.01-m accuracy) from each trap location to the nearest shrub (> 0.5 m tall) using a 50-m open-reel measuring tape. We also recorded the species of the nearest shrub. We then determined whether at each trap station, no kangaroo rats, only TKR, only HKR, or both species were captured.

We used Contingency Table Analysis and a Chi-square Test to compare the proportion of stations with Seepweed or Iodine Bush as the nearest shrub between plots, the proportion of stations with TKR and HKR captures between plots, the proportions of stations with captures of TKR and of both species, and the proportion of stations with captures of TKR, HKR, or both species relative to the species of shrub nearest the station. For the last two analyses, data from the two plots were combined as there were insufficient data to conduct the analyses for each plot. We used a Yates correction for 2 × 2 Contingency Tables (Zar 1984). We used a Mann-Whitney U test to compare the mean distance from stations to the nearest shrub between plots and to compare the mean distance to the nearest shrub between stations with TKR and HKR captures. We also used a Levene statistic to compare variances in distance to the nearest shrub between stations with TKR and HKR captures. For all statistical analyses, we set α at 0.10. We chose a more relaxed alpha value in an effort to reveal potential ecological relationships that could be more fully explored through further investigation (Taylor and Gerrodette 1993; di Stefano 2003; Scherer and Tracey 2011). We analyzed data using the SPSS statistical software package (ver. 26, International Business Machines Corporation, Armonk, New York).

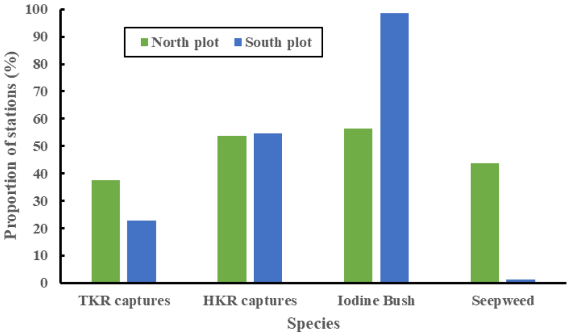

Figure 3. Proportions of stations with captures of Tipton Kangaroo Rats (*Dipodomys n. nitratoides)* and Heermann's Kangaroo Rats (*D. heermanni*) and the proportions at which either Iodine Bush or Seepweed were the nearest shrub on two plots on the Unit 15 study area, Kern National Wildlife Refuge, California, October 2018.

Results

The species of shrub nearest to each trap station differed between plots ($\chi^2 = 36.723$, df = 1, $P < 0.001$; Fig. 3). The proportion of stations with Seepweed was higher on the north plot (43.7%) than on the south plot (1.3%). The proportion of stations with Iodine Bush was higher on the south plot (98.7%) than on the north plot (56.3%). The mean (± standard error) distance from stations to the nearest shrub was not significantly different ($U = 3013.0$, $P = 0.963$) between the north plot (4.88 ± 0.95 m) and the south plot (4.20 ± 0.78 m) indicating that shrub density was similar between the plots.

For TKR, we caught 26 individuals (16.3/100 trap-nights) on the north plot and 16 (10.0/100 trap-nights) on the south plot. For HKR, we captured 36 individuals (22.5/100 trap-nights) on the north plot and 41 (25.6/100 trap-nights) on the south plot. The proportion of stations with any TKR captures was significantly higher ($\chi^2 = 3.362$, df = 1, $P = 0.067$; Fig. 3) on the north plot (37.5%) than on the south plot (22.7%). The proportion of stations with HKR captures was not significantly different ($\chi^2 < 0.013$, df = 1, $P = 0.909$; Fig. 2) between the north plot (53.8%) and the south plot (54.7%).

The proportion of stations where we only caught TKR (28.8%) was significantly higher ($\chi^2 = 10.63$, df = 1, $P = 0.001$) than the proportion where we caught both TKR and HKR (11.0%). The proportion of stations with captures of only TKR, only HKR, or both species did not differ significantly ($\chi^2 = 0.887$, df = 2, $P = 0.642$) based on the species of the nearest shrub (Fig. 4). The

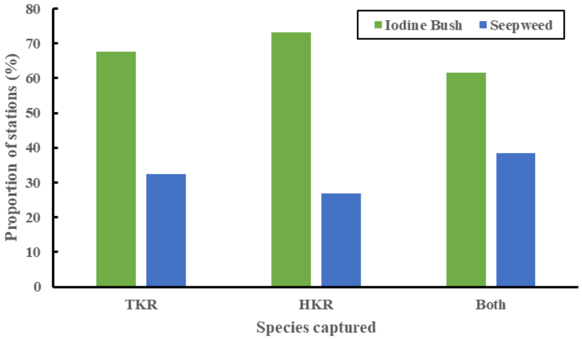

Figure 4. Proportions of stations with captures of Tipton Kangaroo Rats (*Dipodomys n. nitratoides*) and Heermann's Kangaroo Rats (*D. heermanni*), or both species relative to the nearest shrub species on the Unit 15 study area, Kern National Wildlife Refuge, California, October 2018.

mean distance between trap stations and the nearest shrub did not differ significantly ($U = 3182.5$, $P = 0.317$) between TKR (2.57 ± 0.44 m) and HKR (4.53 ± 0.85 m); however, the variance in distances was significantly greater (Levene statistic = 6.95; df = 129; $P = 0.009$) for HKR (60.84) than for TKR (8.96).

Discussion

Based on our trapping results on Unit 15 at KNWR, we did not detect apparent avoidance of shrubs by TKR when compared to HKR. We predicted that TKR would be found farther from shrubs because HKR, a larger intraguild competitor, might be more prevalent near shrubs. TKR capture locations generally were closer to shrubs than those for HKR, although the difference was not statistically significant due to the high variability in HKR distances. No TKR capture location was > 14.5 m from a shrub whereas measured HKR capture locations were as far as 40.6 m from a shrub. Also, we captured HKR at two locations that were > 50 m from the nearest shrub (the exact distance was not measured). Thus, HKR clearly were traveling farther from shrubs than TKR.

The smaller proportion of stations with captures of both kangaroo rat species compared to the proportion with only TKR captures suggests some avoidance of HKR by TKR. Tennant and Germano (2013) documented a 500% increase in TKR on a plot from which HKR had been removed whereas no increase in TKR was observed on an associated control plot from which HKR were not removed. Cypher et al. (2017, *op. cit.*) reported that HKR were not detected on 46.7% of sites where TKR were captured, and capture rates for the two species were inversely related on sites where both species were present. So, HKR clearly can have an adverse competitive effect on TKR. Competitive suppression of smaller kangaroo rats by larger ones has been well documented (e.g., Blaustein and Risser 1976; Frye 1983; Brown and Munger 1985; Reichman and Price 1993; Perri and Randall 1999). Such suppression has been assumed to adversely impact TKR and other subspecies of San Joaquin kangaroo rats as well (Williams and Germano 1992).

Shrubs potentially facilitate coexistence between the two kangaroo rat species by providing escape cover for TKR if they are harassed by HKR. The presence of shrubs increases habitat complexity and this can facilitate sympatric coexistence by two competing desert rodent species (sensu Rosenzweig and Winakur 1969). On the Carrizo Plain, the Giant Kangaroo Rat (*D. ingens*), a relatively large species (131–180 g; Jameson and Peeters 1988), is abundant and wide-spread. Consequently, the Short-nosed Kangaroo Rat (comparable in size to TKR) is competitively excluded from areas without shrubs but is found in some areas where shrubs are present (Williams 1992). Avoidance may be the behavioral mechanism that allows a smaller kangaroo rat species to coexist with a larger one (Perri and Randall 1999; Tennant and Germano 2013), and shrubs may facilitate avoidance of HKR by TKR.

In a previous study on Unit 15, in which TKR with transmitters were consumed by Pacific Rattlesnakes (*Crotalus oreganus*), I found that the snakes used shrubs for daytime cover (Brian Cypher, unpubl. data). Thus, I speculated that shrubs potentially constitute sites of elevated predation risk for TKR. Interestingly, Bouskila (1995) found that the Merriam's Kangaroo Rats (*D. merriami*), which are similar in size to TKR, exhibited preferential use of microhabitats with shrubs, despite greater snake predation risk, because these microhabitats were avoided by Desert Kangaroo Rats (*D. deserti*), a larger intraguild competitor. The lack of avoidance of shrubs by TKR on Unit 15 may indicate that a similar dynamic might be occurring where competition from HKR is much stronger than the risk of predation from snakes.

TKR capture locations on the Unit 15 plots appeared random with respect to the nearest shrub species. Thus, any shrub may serve as cover for TKR; however, TKR were more abundant on the north plot where Seepweed was more abundant. TKR have been consistently more abundant on this plot over the 4 y that population monitoring has been conducted (KNWR, unpubl. data). Cypher et al. (2017, *op. cit.*) found that Seepweed tended to be consistently present on sites where TKR were detected. Thus, an association between TKR and Seepweed may exist, but on a larger scale such as a study site or landscape and not on a microsite (e.g., capture location) scale. A potential explanation is that Seepweed is an indicator of site conditions (e.g., soil composition and texture, relative soil moisture and saturation, etc.) that are more suitable for TKR. HKR abundance was similar on the two plots despite the difference in shrub composition. The HKR is more of a generalist species (Kelt 1988) and is found in a wider range of environmental conditions compared to TKR.

In summary, this and previous studies suggest that shrubs may facilitate the occurrence of TKR in areas where HKR, a larger intraguild competitor, also occurs. Shrub density did not appear to affect TKR abundance on our study site. Seepweed may positively influence the presence and abundance of TKR at a given site but does not appear to influence microhabitat use on the site as determined by trapping. Seepweed may be more of an indicator of site conditions associated with suitable habitat for TKR. Our results, however, were based on a relatively small data set collected on one site in one year but can serve as hypotheses for more extensive investigation, preferably employing more rigorous techniques such as radio telemetry.

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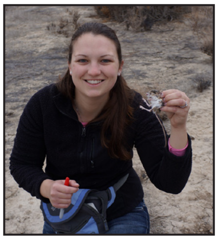

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STRATEGIES FOR TRANSLOCATING ENDANGERED GIANT KANGAROO RATS (*DIPODOMYS INGENS*)

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***Abstract*.—The Giant Kangaroo Rat (*Dipodomys ingens*: GKR) is an imperiled species in the San Joaquin Desert of California due to profound habitat loss. Occupied habitat is still being developed and translocation is becoming a common mitigation strategy. In 2012 and 2013, we translocated GKR from proposed oil-well pad sites in western Kern County, California, to a nearby conservation area. In 2012, we semi hard-released 43 animals into artificial burrows and in 2013, we soft-released 38 animals into artificial burrows within enclosures. The soft-released GKR exhibited higher apparent survival based on subsequent live-trapping. Retention time in enclosures did not affect survival, probably because GKR rapidly burrowed out of enclosures. Both hard-released and soft-released GKR exhibited rapid fidelity to their release sites. Thus, the enclosures primarily may have afforded translocated animals protection from predators while they acclimated to their new surroundings. Soft-released GKR translocated as a social group exhibited higher survival than those translocated indiscriminately. A GKR population has persisted at the release site and were still present as of April 2020. Based on our results, we recommend that GKR be translocated in social groups and soft-released in suitable habitat on conservation lands.**

Key Words.—California; hard release; soft release; survival; well pad

INTRODUCTION

The San Joaquin Desert of California is home to many species of conservation concern (U.S. Fish and Wildlife Service [USFWS] 1998; Germano et al. 2011). Urban, industrial, and agricultural development is still occurring in this region, and federally and state-listed species are sometimes present on sites planned for such development. An increasingly common mitigation strategy is to translocate rare animals and plants off of development sites, usually to designated conservation areas. Translocation has well-documented risks (e.g., Griffith et al. 1989; Dickens et al. 2010; International Union for Conservation of Nature 2013) and many efforts are not successful (Fischer and Lindenmayer 2000; Germano 2001; Armstrong and Seddon 2008). In some cases, the results of translocations are not even monitored (Tennant et al. 2013). Thus, any efforts to assess and improve translocation strategies are valuable.

Translocation efforts have been conducted for several kangaroo rat (*Dipodomys* spp.) species with limited success. These species include the Tipton Kangaroo Rat (*D. nitratoides nitratoides*; Federally listed Endangered, California listed Endangered), Stephen's Kangaroo Rat (*D. stephensi*; Federally listed Endangered, California listed Threatened), and the common and unlisted Heermann's Kangaroo Rat (*D. heermanni*). These translocation efforts have included both soft releases, in which individuals are confined to the release site for some period of time, and hard releases, in which individuals are not confined to the release site. To date, neither strategy has proven superior to the other. Previous kangaroo rat translocation attempts were thoroughly reviewed by Germano (2001, 2010), Shier and Swaisgood (2012), Tennant et al. (2013), and Tennant and Germano (2017).

The Giant Kangaroo Rat (*Dipodomys ingens*; GKR) is endemic to the San Joaquin Desert (Williams and Kilburn 1991; Germano et al. 2011). The GKR is the largest kangaroo rat (Williams et al. 1993). Each GKR inhabits an extensive burrow system referred to as a precinct. They are larder hoarders and store large quantities (up to multiple liters) of seeds, their primary food, in subterranean chambers within the precinct. GKR exhibit high fidelity to their precinct and will vigorously defend it. Thus, precincts constitute a critical aspect of GKR ecology (Williams and Kilburn 1991).

GKR are listed federally and by the state as endangered, primarily due to profound habitat loss and degradation (USFWS 1998). They primarily persist in three large and three small populations (USFWS 1998). The large population areas are considered to be core areas and are critical for the conservation and recovery of GKR. One of these core areas is in western Kern County, which also is a region of extensive hydrocarbon (crude oil and natural gas) production. New facilities and infrastructure (e.g., well pads, pipelines, roads) are sometimes constructed in habitat occupied by GKR. When this happens, an increasingly common mitigation strategy is to translocate the GKR to another area, preferably one that is conserved and not under threat of future development.

In 2012 and 2013, the lead author coordinated the translocation of 81 GKR from well-pad construction sites to conservation lands approximately 14 km away. The animals translocated in 2012 were semi hard released (i.e., released into artificial burrows provisioned with seed). The animals translocated in 2013 were soft released and we hypothesized that this strategy would result in better survival, as suggested by Germano et al.

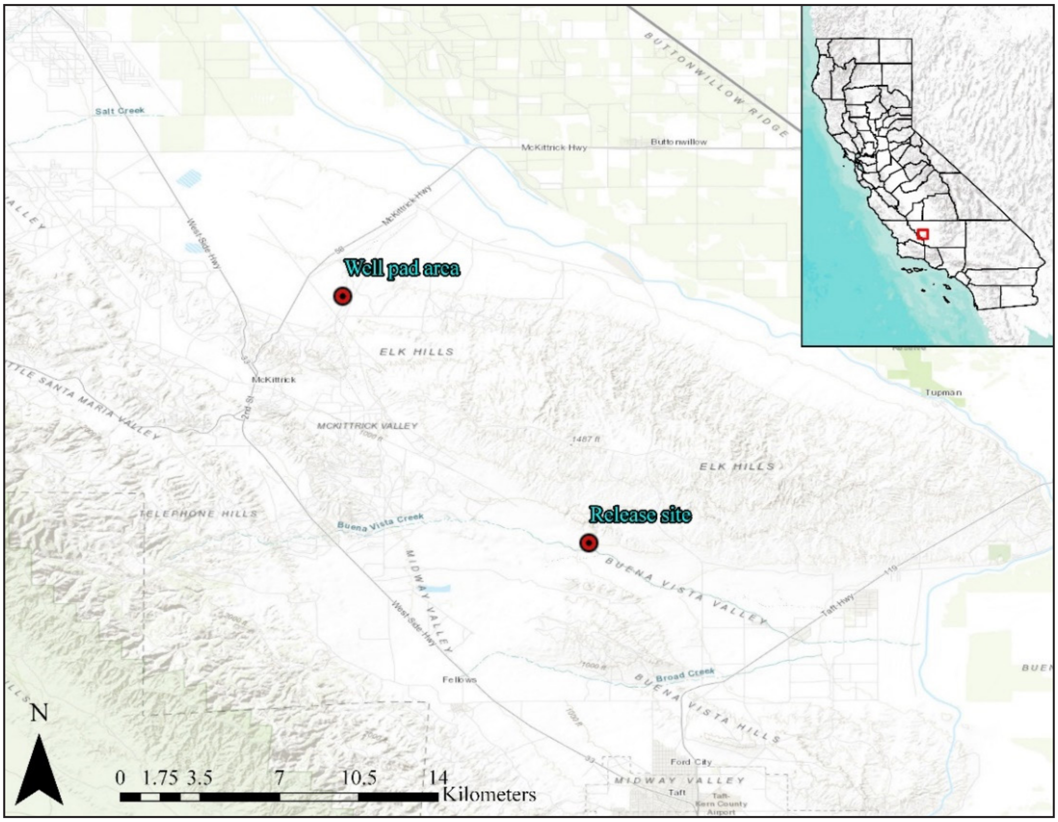

Figure 1. Locations of well pads from which Giant Kangaroo Rats (*Dipodomys ingens*) were translocated and the release site in Kern County, California.

(2013) and Tennant et al. (2013). Furthermore, in 2013 we compared survival of animals moved as a social group to those not moved as a social group as we hypothesized that this might improve survival as well. Our objective was to compare translocation strategies with an overall goal of establishing a GKR population at the release site. We predict that soft-released GKR will exhibit higher survival than hard-released individuals and that GKR moved in social groupings would have higher survival rates than those moved irrespective of trapped location.

Methods

Study areas.—The well pad sites from which we translocated GKR were in the Gunslinger Unit of the Occidental of Elk Hills (OEH) oilfield, and were located approximately 4 km northeast of McKittrick, Kern County, California (Fig. 1). The release site was located approximately 14 km southeast of the well pad sites on OEH conservation lands in the Buena Vista Valley (Fig. 1). We selected this release site because it was not occupied by GKR at the time but was within a region with suitable habitat where the specie was known to consistently occur (USFWS 1998). Habitat conditions in the well pad site and release site were similar. The terrain in both areas was flat to gently rolling and the elevation was approximately 100 m. The regional climate was Mediterranean in nature and was characterized by hot, dry summers, and cool, wet winters with frequent fog. Mean maximum and minimum temperatures were 35° C and 18° C, respectively, in summer, and 17° C and 5° C, respectively, in winter (https://wrcc.dri.edu/cgi-bin/cliMAIN.pl?ca8752). Annual precipitation averaged 137 mm and occurred primarily as rain falling between October and April (https://wrcc.dri.edu/cgi-bin/cliMAIN.pl?ca8752.). The vegetation community at both sites was characterized as Lower Sonoran Grassland (Twisselmann 1967) or Allscale Series (Sawyer and Keeler-Wolf 1995). The plant community consisted of arid shrublands dominated by Desert Saltbush (*Atriplex polycarpa*). Ground cover consisted primarily of annual grasses and forbs and was dominated by Red Brome (*Bromus rubens madritensis*) and Red-stemmed Filaree (*Erodium cicutarium*).

GKR capture, translocation, and monitoring.—We live-trapped GKR on the well pad sites by setting traps near kangaroo rat burrows that appeared active. We set traps at burrows within the enclosed (metal flashing) well pad site. We used Sherman XLK Extra-Large Kangaroo Rat Traps (30.5 × 9.5 × 7.6 cm; H.B. Sherman Traps Inc., Tallahassee, Florida). We opened the traps within 2 h of sunset and we provisioned each trap with a handful of millet seed and two sheets of crumpled, unbleached paper towels for insulation and to keep kangaroo rats from chewing on the traps. We checked and closed the traps the following morning within 2 h of sunrise. After checking that we had captured a GKR, we placed them

Figure 2. Soil auger being used to create artificial burrows for translocated Giant Kangaroo Rats (*Dipodomys ingens*) as a release site in western Kern County, California. (Photographed by Brian Berry).

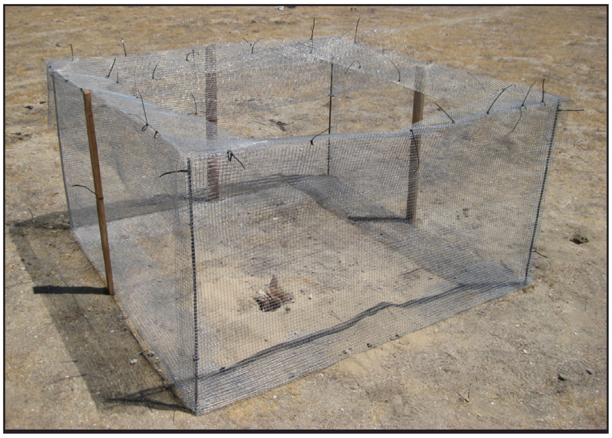

Figure 3. Soft-release enclosure for translocated Giant Kangaroo Rats (*Dipodomys ingens*) at a release site in western Kern County, California. (Photographed by Larry Saslaw).

back in the trap and then transported them via vehicle to the release site or to a home office for fitting a radio transmitter and then subsequent release at the study site.

In 2012, we used a semi hard-release approach for all captured GKR. At the release site, we created artificial burrows using an 8.5-cm soil auger angled to a depth of 60–90 cm below the surface and about 120 cm in length (Fig. 2). The artificial burrows were located at least 10 m apart and no resident GKR were present at the site. We provisioned each burrow with about 250 ml of birdseed. During trapping conducted 4–9 April 2012 at the well pad sites, we captured 43 GKR (22 females, 21 males) and transported them directly to the release site. We then released one animal into each artificial burrow and loosely plugged the entrance with paper towels to discourage animals from immediately leaving the burrows (Germano et al. 2013). The GKR could exit the artificial burrow at will by simply pushing through the paper towel, which we confirmed the next day. Prior to release, we attached a uniquely numbered No.1 Monel ear tag (American Band and Tag Co., Newport, Kentucky) in each ear of an individual. We placed animals in the burrows in a pattern that roughly approximated that of the capture locations at the well pad sites. Thus, animals captured closer together were released closer together and animals captured farther apart were released farther apart (social grouping). We did not have a control group in 2012 and so could not evaluate whether translocating with social grouping improved survival. Also, comparing the 2012 data to 2013 would be confounded by the inclusion of soft releases in 2013.

In 2012, we trapped kangaroo rats at the release site 35 d post-release (for four nights from 14–18 May). Trapping methods were similar to those used to capture animals at the well pad sites. We placed traps near the artificial burrows as well as nearby locations with active kangaroo rat sign. We recorded the ear tag number, weight, and reproductive condition for all GKR captured and we recorded the capture location using a Global Positioning System (GPS) unit. We re-trapped the site approximately 5 mo later (16–18 October 2012) to further assess GKR survival and movements.

In 2013, we used a soft-release approach for GKR we translocated. At the release site, we constructed artificial burrows as in the 2012 release; however, we also constructed over each burrow an enclosure using 1-cm hardware cloth (Fig. 3). Each enclosure was either 183 × 183 × 92 cm in length, width, and height, or 244 × 183 × 92 cm (some just happened to be built larger, but the difference in sizes was not considered sufficiently significant to affect GKR survival). The enclosure was held in place by 122-cm long pieces of 1-cm diameter rebar driven into the ground at each corner and a 122-cm long wood lath driven into the ground on each side. On all sides of the enclosure, a 30-cm flange extended inward along the surface of the ground at the bottom of the enclosure to inhibit GKR from quickly leaving the enclosure by digging underneath the side (Fig. 3). We staked this flange to the ground so that it remained flat. At each corner, we formed a 7 cm fold on one end of each side piece to seal the corner. Additionally, we folded a 60-cm wide piece of hardware cloth in half at 90° that we attached inward around the top edge of each enclosure to discourage GKR from climbing up and over the side (Fig. 3). Finally, we extended chicken wire across the top of the enclosure to exclude entry by avian predators. We attached the sides of the enclosure to the rebar supports and the chicken wire top with 30-cm nylon cable ties. Finally, a tight seal around the bottom of the enclosure was formed by staking the bottom edge to the ground with 30-cm spikes. We located enclosures avoiding any small mammal burrows and we spaced them at least 10 m apart (Fig. 4) and at least 10 m from any 2012-released GKR precincts.

During trapping we conducted 11–18 June 2013 at the two well pad sites, we captured 36 GKR (24 females,

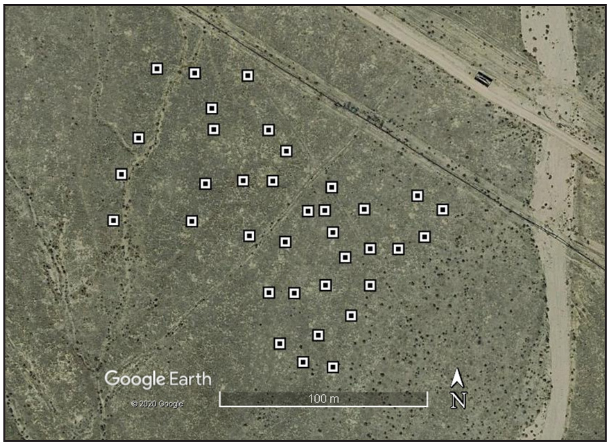

Figure 4. Aerial image of release pen locations for translocated Giant Kangaroo Rats (*Dipodomys ingens*) at a release site in western Kern County, California. (Aerial image from Google Earth).

12 males). We captured two additional male GKR at two well pad sites 28 June and 8 July 2013. We marked all GKR translocated in 2013 with a passive integrated transponder (PIT) tag inserted subcutaneously in the shoulder region of the back (Williams et al. 1997). Additionally, we fitted 19 (11 females, eight males) GKR with radio transmitters (Model BD-2, Holohil Systems, Carp, Ontario, Canada) attached to beaded chain collars to monitor survival and movements. We maintained these GKR in 19-l buckets for 5 d at a home office to ensure a proper collar fit. During 13–17 June 2013, we transported and released 36 GKR to the release site and we placed each one in an artificial burrow within the enclosures. We then secured enclosures with cable ties to prevent entrance or exit from the cage. We released all of the 19 radio-collared GKR into their respective artificial burrows and enclosures on 17 June. To determine if survival of GKR was affected by retention time in the enclosures, we removed 14 enclosures 22–24 d post-release and the remainder after 33–35 d. Also, we moved 14 GKR from one of the well pad sites as a social group, meaning that they all were released in the same part of the release site and in a pattern that approximated their relative capture locations. Thus, animals captured closer together were released closer together and animals captured farther apart were released farther apart. We did not translocate the other 24 GKR we caught as a social group, and we simply placed them in available burrows at the release site in no particular pattern.

We tracked the radio-collared GKR daily for the first 10 d and several times per week thereafter using a hand-held receiver and 3-element Yagi antenna. Tracking either led to a burrow that the animal was in or to a mortality site where typically we only found the transmitter. In either case, we recorded the location with a GPS unit. Also, using methods similar to those used in 2012, we trapped at the release site approximately four weeks, six weeks, and 10 weeks post release. During the last trapping session, we removed radio collars from collared GKR.

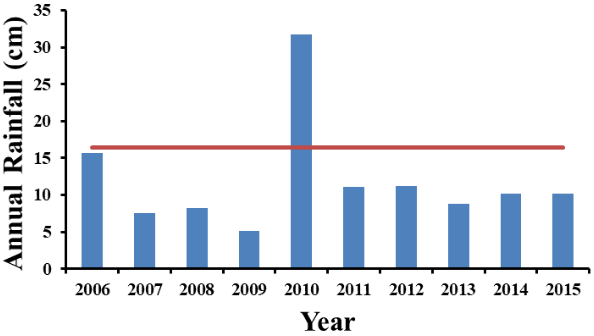

Figure 5. Annual precipitation at Bakersfield, California, from 2006 to 2015. The horizontal line is the long-term average from 1889 to 2019. (https://weather.gov/hnx/bflmain).

We trapped at the release site again in March 2014 to determine if any GKR were still present. We placed two traps at each artificial burrow with obvious GKR activity (e.g., fresh digging) and at natural burrows in the area exhibiting possible GKR activity. Due to successive years of low precipitation and concomitant plant growth in spring 2014 (Fig. 5), we distributed approximately 250-500 ml of bird seed around active burrows to enhance GKR survival during this period of low food availability. We distributed seed approximately every two weeks from March through the end of 2014. We trapped on the release site again in April 2015 to assess the status of GKR.

Statistical analyses.—We used Contingency Table Analysis and a Chi-square Test to compare the proportion of translocated GKR detected after 40 d in 2012 to the proportions of GKR detected after 30 d and after 60 d in 2013. We used the same analysis to compare the proportions of animals detected after 60 d in 2013 between animals whose enclosures had been removed at 22–24 d to those whose enclosures had been removed at 33–35 d. All of the Chi-square Tests entailed 2 × 2 Contingency Tables and so we used a Yates correction for all tests (Zar 1984). We used a Fisher exact text (due to small cell sample sizes) to compare the proportion of GKR that were known to be alive after 60 d in 2013 between animals moved as a social group and those not moved as a social group. For all statistical analyses, we considered significance at $\alpha = 0.10$. We chose a more relaxed alpha value in an effort to reduce the risk of committing a Type II error and not detecting a potentially useful conservation strategy (Taylor and Gerrodette 1993; di Stefano 2003; Scherer and Tracey 2011).

Results

In 2012, 40 d post-release, we caught 14 of the 43 (32.5%) GKR that we translocated. Of these 14, we captured nine within 20 m of the artificial burrow into which we released them. After approximately 6 mo post-release, we caught five of the 43 (11.6%) translocated

GKR. In 2013, 30 d post-release, we caught 24 of 38 (63.2%) translocated GKR, and approximately 60 d post-release, we caught 20 of the 38 (52.6%). Of these 20, we captured 14 within 5 m of the artificial burrow into which we released them.

The difference in the proportion of translocated GKR detected after 40 d in 2012 (32.5%) and detected after 30 d in 2013 (63.2%) was significant ($\chi^2 = 6.41$, df = 1, $P = 0.011$), although some number of animals still were in the enclosures (the exact number is not known because some GKR apparently dug out before we removed the enclosures). The difference in the proportion of translocated GKR detected after 40 d in 2012 (32.5%) and those detected after 60 d in 2013 (52.6%) was not significant ($\chi^2 = 2.56$, df = 1, $P = 0.110$). The proportion of animals detected after 60 d in 2013 did not differ significantly ($\chi^2 = 0.08$, df = 1, $P = 0.777$) whether their enclosures had been removed at 22–24 d (10/13 = 76.0%) or 33–35 d (10/18 = 55.5%). Also, the proportion of GKR that were moved as a social group that were known to be alive after 60 d (10/14 = 71.4%) was significantly higher (Fisher exact text; $P = 0.075$) than the proportion that was not moved as a social group (10/24 = 41.7%).

The ultimate fate of the GKR translocated in 2012 is not known. Of the 38 GKR we followed in 2013, four were never detected post-release, 25 were still alive after 30 d and 20 of those were still alive after 60 d. For the remaining nine animals we moved in 2013, one was killed by a predator while in a trap, we found one dead on the ground outside of its enclosure, two dead in burrows, and just the radio collar of the last five GKR lying on the ground 22 m, 109 m, 115 m, 120 m, and 142 m from their release locations. We think these last five were killed by a predator, probably an owl.

In 2014, we captured 14 GKR in two nights of trapping, none of which were from the 2012 release but nine of which were marked animals from the 2013 translocation. Thus, 23.7% (9/38) of the translocated GKR in 2013 had survived for 9 mo post-release. Based on weight (< 100 g) and pelage characteristics (Williams and Kilburn 1991), three of the six new GKR were young-of-the-year. Approximately 22 mo after the 2013 releases, we caught 17 GKR in 2015. We captured one ear-tagged GKR from the 2012 translocation and one PIT tagged GKR from the 2013 translocation and three of the unmarked animals were young-of-the-year.

DISCUSSION

Although our monitoring methods were not identical between years, survival of translocated GKR apparently was higher in 2013 when we used a soft-release strategy compared to 2012 when we used a semi hard-release strategy. This was consistent with our prediction that soft-released animals would exhibit higher survival. Prior to our study, GKR had been translocated in other locations, but either soft-release or hard-release were used exclusively. In July 1989, GKR were translocated and hard released into artificial burrows at two sites on the Carrizo Plain in San Luis Obispo County (Williams et al. 1993). At one site, 15 of 30 (50%) translocated animals were recaptured approximately 1-mo post-release. At the other site, 12 of 30 (40.0%) animals were recaptured 11 mo post-release and at least one translocated GKR was captured each month thereafter through November 1991 (28 mo post-release). These values are generally similar to those observed in our study.

From September 2011 to August 2013, 221 GKR also were translocated at the California Valley Solar Ranch in San Luis Obispo County, California, as part of a solar farm project (H.T. Harvey, unpubl. report). The animals were soft-released into artificial burrows within large enclosures (6 × 3 × 1.2 m length, width, height) constructed with 1.3-cm mesh hardware cloth and covered with shade cloth to exclude aerial predators. The enclosures were left in place until no activity was observed for at least 48 d or until it was clear that the GKR inside had expanded its burrow to outside of the enclosure. Survival was not specifically assessed, but using a passive PIT-tag reader system, 63 of the 221 (28.5%) of the translocated GKR were still present at their release sites 47 d post-release and one was still present after 721 d. In comparison, in our study 32.5% of translocated GKR were still present after 40 d in 2012 and 52.6% were still present after 60 d in 2013.

In two translocation efforts conducted with other kangaroo rat species from the San Joaquin Desert, soft-release and hard-release strategies were compared directly, but the results were equivocal. In 2006, 144 endangered Tipton Kangaroo Rats (*D. nitratoides nitratoides*) were translocated and 86 were soft released while 36 were hard released. Based on animals with radio collars, survival to 30 d was 58.3% for soft-released animals and 37.5% for hard-released animals, although these values were not significantly different (at 0.05 α; Germano et al. 2013). In 2009, 43 Heermann's Kangaroo Rats (*D. heermanni*) were translocated and 32 were soft-released while 10 were hard-released (one escaped prior to release). Based on radio-collared individuals, survival actually trended higher among the hard-released individuals although the values were not significantly different (Tennant and Germano 2017).

In southern California, translocations of the endangered Stephen's Kangaroo Rats (*D. stephensi*), efforts employing soft release were more successful than those employing hard release, although the efforts differed in a number of regards. Stephen's Kangaroo Rats were translocated and hard released in two efforts in 1992 and 2002 (O'Farrell 1984; Spencer 2003 cited in Shier and Swaisgood 2012). In the first effort, none of the 599 translocated animals could be found after 11 mo. In the second, 40% were still present after 4 mo, but none could be found after one year. In another attempt with this species, 54 animals were translocated in 2008 and another 45 in 2009. All were soft released into enclosures that were removed after one week. Early survival estimates were not given, but a thriving, expanding population was reported to be present on the release site 3 y post-release (Shier and Swaisgood 2012). Success was attributed to soft release along with social group translocation. Finally, in a translocation of endangered San Bernardino Kangaroo Rats (*D. merriami parvus*) in southern California, the animals were hard released and no artificial burrows were provided. Four months post-release, 40% of the 15 translocated animals were still present and most were reproductively active (O'Farrell 1999).

In our study, retention time in the enclosures did not affect the survival of translocated GKR contrary with our prediction that a longer retention time would result in higher survival. This may have been due to the relatively small difference between the two retention times (22–24 d versus 33–35 d). The intent of the different retention times was to determine whether GKR would be more likely to exhibit burrow fidelity if they were confined to the release site for a longer period; however, GKR have such a strong affinity for where they are that confinement may not be necessary. As is common among the larger kangaroo rat species (e.g., Bannertail Kangaroo Rat, *D. spectabilis*; Reichman et al. 1985), individuals construct elaborate burrow systems, store huge quantities of food in these systems (i.e., larder-hoard), and limit their activity to just one system. Thus, this system is integral to their survival and fitness, and consequently, translocated GKR seem to rapidly adopt a burrow and begin modifying and expanding it.

We observed modification and expansion of the artificial burrows at most of the release sites, including those of the semi hard-released animals. At some of the release sites, these modifications were evident within 24 h of release. Rapid, and in many cases immediate, modifications also were observed among the soft-released GKR at the California Valley Solar Ranch (H.T. Harvey, unpubl. report). In the 1989 Carrizo Plain translocations where all of the GKR were semi hard released, observations on the first night after release indicated that GKR exited artificial burrows, explored the immediate area up to 50 m, and then quickly returned to the burrow (Williams et al. 1993). By the next day, the GKR had clearly begun to modify many of the artificial burrows. We captured 64.3% of the semi hard released GKR within 20 m of their release site 30 d post-release and 70.0% of the soft-released GKR within 20 m of their release site 60 d post-release. Thus, confinement may not be necessary for GKR to rapidly develop fidelity to the release site. Results regarding soft versus hard releases of smaller kangaroo rat species may be equivocal in part because they have less fidelity to a particular burrow system and instead, consistent with their scatter-hoarding behavior, commonly use multiple burrows distributed over a larger area than that used by GKR (Reichman 1983; Tennant and Germano 2013).

Enclosures may be beneficial in reducing predation on newly translocated kangaroo rats. GKR, and indeed all kangaroo rats, are prey for a multitude of predators. Thus, they naturally will have relatively high mortality rates just from natural predation. Translocated animals are particularly vulnerable, especially in the early days following release, because they are unfamiliar with their new environment and also may be somewhat disoriented due to the stress of being trapped and transported (Banks et al. 2002; Hamilton 2010). High predation rates were strongly suspected of contributing to the failure of a GKR translocation at one site on Carrizo Plain in 1989 because the release site was unintentionally placed within the home range of a pair of San Joaquin Kit Foxes (*Vulpes macrotis mutica*; Williams et al. 1993). High predation rates also were reported for semi hard released (no cages) Tipton and Heermann's Kangaroo Rats within the first few days following release (Germano 2010; Germano et al. 2013). Enclosures may enhance survival of translocated animals by affording them protection from predators while they are acclimating to their new environs and creating a suitable burrow system. Even GKR that dug out of enclosures were found to continue returning to and using the burrows within the enclosures in this study as well as at the translocation at the California Valley Solar Ranch (H.T. Harvey, unpubl. report). The protection from predators afforded by the enclosures likely explains the higher apparent short-term survival of soft-released GKR compared to hard released or semi hard released GKR in our study.

Maintaining social grouping of GKR during this translocation resulted in higher post-release survival, which was consistent with our prediction. Shier and Swaisgood (2012) compared groups of Stephen's Kangaroo Rats translocated with and without neighbors. Those translocated with neighbors exhibited significantly higher survival, site fidelity, and reproductive success. Individuals translocated with neighbors spent less time fighting with neighbors and more time foraging and creating new burrows. Similar positive results were found for Black-tailed Prairie Dogs (*Cynomys ludovicianus*) translocated in family groups in New Mexico (Shier 2006). Although GKR are essentially solitary, neighbor recognition and familiarity apparently enhance fitness (Randall et al. 2002).

We seem to have established a population of GKR at our release site, which was a primary goal of the translocation effort. Based on the trapping conducted at the site 9 mo and 22 mo after the 2013 releases, GKR were still present, including some of the translocated animals. The number may have been higher at 9 mo, but trapping was terminated after two nights due to trap disturbance by San Joaquin Kit Foxes or Coyotes (*Canis latrans*). The presence of unmarked animals, and particularly the capture of juveniles, indicated that reproduction was occurring at the site. GKR sign (e.g., large burrow entrances, vertical burrow entrances, large scat) was abundant at the release site and surrounding area during site visits in July 2017 and April 2020. This was particularly encouraging given the region-wide low annual precipitation in 2012–2015 (Fig. 5) and concomitant marked declines in GKR abundance recorded on the Carrizo Plain (Prugh et al. 2018) and the Lokern Natural Area (Germano and Saslaw 2017; Greg Warrick, unpubl. data). Six-month survival of GKR at the Lokern study site located 23 km to the north was 29.7% from April to October in 2012 and 12.7% from April to October in 2013 (Germano and Saslaw 2017). In our study, 9-mo survival from June 2013 to March 2014 was 23.7%. Overwinter survival between summer 2013 to spring 2014 on the Carrizo Plains was between 10% and 20% (Prugh et al. 2018). Thus, we considered our translocation effort to be a success.

Our enclosure design appeared to be effective because it prevented GKR from immediately vacating the release site while also affording them protection from predators while they acclimated to their new environment. The design we used was relatively simple, easy to construct as well as remove, and relatively inexpensive (approximately $142 per enclosure for materials). Thus, enclosure designs need not be complex or expensive to be effective. Also, the bottom edge of our enclosures had a flange of hardware cloth and was not buried. This precluded the need for trenching or digging. Not only did this reduce the labor needed to install the enclosures but it also significantly reduced the potential for impacts to endangered Blunt-nosed Leopard Lizards (*Gambelia sila*). This species occurs almost everywhere that GKR occur and also use burrows, including kangaroo rat burrows (USFWS 1998). Ground-disturbing activities can result in injury or death of Blunt-nosed Leopard Lizards.

Our comparison of hard-release and soft-release strategies was not ideal in that the efforts were conducted in different years and differences in annual environmental conditions (e.g., precipitation, seed production, predator abundance) could have influenced results. Despite these potential weaknesses, ours is the only effort to date that provides a quantitative comparison between the release strategies as applied to GKR. Clearly, further research would be informative as the need for future translocations is likely given the continuing development activities in GKR habitat.

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NOTES

Bighorn Sheep (*Ovis canadensis*) Ram Associating with a Herd of Aoudad (*Ammotragus lervia*) at Big Bend Ranch State Park, Texas

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***Abstract*.—The native North American Bighorn Sheep (*Ovis canadensis*) and the non-native Aoudad (*Ammotragus lervia*) are known to compete for the same resources and habitat, and they have been recognized to be behaviorally incompatible. At Big Bend Ranch State Park, Texas, a long-term study using camera-traps produced images of an incident that is contradictory to expected behaviors between these two species. Camera-trap images depict an adult male Bighorn Sheep within a group of Aoudad. Behaviors were passive and tolerant among female and immature Aoudad and the male Bighorn Sheep.**

Key Words.—camera-traps; behavior; Chihuahuan Desert; social interaction; Trans-Pecos

Bighorn Sheep (*Ovis canadensis*) are distributed throughout mountainous regions in western North America from British Columbia southward to Baja and northern Mexico eastward to the Trans-Pecos region of Texas (Shackleton 1985; Festa-Bianchet 1999). In Texas, populations of the Desert Bighorn Sheep (*O. c. mexicana*), a subspecies adapted to the arid mountain ranges of the Trans-Pecos (Stangl et al. 1994), sustained steep declines through the late 1800s from continuous hunting by railroad crews and mining operations to a point where the Texas State Legislature prohibited all hunting of the species in 1903 (Hailey 1977; Winkler 1977; Kilpatrick 1982). The last of original native Desert Bighorn Sheep in Texas, however, were extirpated from the Sierra Diablo Wildlife Management Area (SDWMA) in 1960 (Hailey 1977; Schmidly 1977; Kilpatrick 1982; Brewer and Hobson 2000). The state of Texas began a reintroduction program during the late 1950s at Black Gap Wildlife Management Area (BGWMA) with Bighorn Sheep transplanted from Arizona (Kilpatrick 1975, 1982; Brewer and Hobson 2000). Once established at BGWMA, reintroductions were made back into SDWMA (Winkler 1977; Kilpatrick 1982), and over the next 40 y, management programs of the Texas Parks and Wildlife Department (TPWD) translocated Bighorn Sheep back into other mountain ranges where the native subspecies had been extirpated, resulting in seven free-ranging populations in the Trans-Pecos region by 2000 (Brewer and Hobson 2000). In 2011, Bighorn Sheep were translocated into the Bofecillos Mountains of Big Bend Ranch State Park (BBRSP) in Presidio County (Hernádez 2013, 2017) for the eighth population in the Trans-Pecos. Although some of the first transplanted Bighorn Sheep from Arizona were from the same subspecies (*O. c. mexicana*; Shackleton 1985) that was extirpated from Texas, other reintroductions comprised sheep from other subspecies. These included *O. c. canadensis* (Rocky Mountain Bighorn) and *O. c. nelsoni*, which are also commonly referred to as Desert Bighorn (Schmidly and Bradley 2016); therefore no subspecific designation is used herein for reintroduced Bighorn Sheep in Texas.

The Aoudad, or Barbary Sheep (*Ammotragus lervia*), is native to the dry mountainous region of northern Africa (Gray and Simpson 1980; Schmidly and Bradley 2016). The species became successfully established in Texas after 44 individuals were released for sport hunting purposes in 1957 and 1958 by TPWD at Palo Duro Canyon State Park in the upper Panhandle (Morrison 1984). Ironically, introductions of Aoudad were initiated as a substitute for diminished hunting availability of Bighorn Sheep. By 1980, free-ranging Aoudad were found across several desert mountain ranges in the Trans-Pecos region and discussions began about control efforts because of the competitive potential of Aoudad with reintroduced Bighorn Sheep (Simpson et al. 1978; Simpson and Krysl 1981; Morrison 1984). In the early 1970s, several Aoudad were released onto private lands that later became parts of BBRSP, however, during a long-term, park-wide survey of mammals, only one individual was sighted (Yancey 1997). Since that work, a 2-y study using remote camera-traps has documented many individuals and sporadic herds of Aoudad throughout BBRSP (Yancey and Manning 2018).

Both Aoudad and Bighorn Sheep typically are gregarious, often forming large herds (Nowak 1999; Krausman and Bowyer 2003). Although there are no accounts of Aoudad herding with other species, groups of Bighorns are known to frequent with Mule Deer (*Odocoileus hemionus*), Elk (*Cervus canadensis*), and Mountain Goats (*Oreamnos americanus*), as well as domestic cattle (*Bos taurus*), sheep (*Ovis aries*), and

Figure 1. Ordered sequence of four of 48 camera-trap images taken 12 August 2016 of an adult male Bighorn Sheep (*Ovis canadensis*) associating with a herd of Aoudad (*Ammotragus lervia*) in the Las Cuevas drainage at Big Bend Ranch State Park, Presidio County, Texas. Images A, B, and C indicate the male Bighorn Sheep in the company of adult female and immature Aoudad. Image D shows the last two male Aoudad in the passing herd. Camera images are horizontally complete although for each of the four images, some non-indicative terrain above and below was cropped to save space.

horses (*Equus caballus*; Smith 1954). We herein report an incident of a male Bighorn Sheep associating with a herd of Aoudad at BBRSP, Texas.

From 1 January 2016 to 26 July 2017, we set camera-traps (model PC8000; Reconyx, Holmen, Wisconsin) at 12 sites in BBRSP. Images of Bighorn Sheep were captured at two sites, whereas images of Aoudad were taken at seven. Both sites where Bighorn images were captured also revealed Aoudad images. On 12 August 2016 from 1503–1506 in the Las Cuevas area of BBRSP (29.494586N, 104.103861W, 1,085 m elevation), a small herd of Aoudad was photographed over 2 min 43 sec. Aoudad were documented moving along a spring-fed creek with 48 total images encompassing 16 triggered series of three-shot bursts at 1 sec intervals. Imbedded within this group of Aoudad was a single adult male Bighorn Sheep (Fig. 1A-C). At the front of the herd, female and immature Aoudad were in view for 1 min 47 sec. Within this period, the adult male Bighorn first appeared and was visible for 15 continuous images covering 38 sec. After the Bighorn moved away from the camera, two adult male Aoudad traversed across the field of view for only 6 sec (Fig. 1D). These male Aoudad were the last images of the photographic sequence.

The male bighorn always was in images with both adult female and immature Aoudad (Fig. 1A-C), but not in images with any identifiable adult male Aoudad. The bighorn was photographed drinking (Fig. 1B) but no Aoudad were noted using water; however, along this stretch of the creek, open pools of water outside the field of view were extensive. During the short period of time covered during these photographs, female and immature Aoudad and the male bighorn seemed relaxed as evidenced by an immature Aoudad grazing near the bighorn (Fig. 1A), and the bighorn drinking water while an immature Aoudad was nearby lying on bedrock (Fig. 1B). At the end of the photograph sequence, all individuals left the camera field of view running up the rocky canyon slope opposite to the camera.

We consider these data a single incident of Bighorn Sheep-Aoudad associations that may be occurring within BBRSP. Because Bighorn Sheep segregate into sexual groups outside of the rut season (Shackleton 1985; Festa-Bianchet 1999), it is possible that other males from a bachelor group may have been out of the field of view as numerous pools of open water occur for over 100 m along this part of the spring-fed creek. Given the wandering nature of rams and their continued interchanging between bands (Jones 1950), however, it is likely that the photographed Bighorn Sheep was an individual that strayed from a bachelor herd.

The gregarious nature of Bighorn Sheep is thought to be mostly the result of predator avoidance instinct (Smith 1954), which possibly stimulated the male bighorn to associate with the Aoudad herd at BBRSP. This behavioral choice would mutually reduce predatory risk and enhance foraging. Traveling with an Aoudad herd could benefit the male bighorn because the largest predator in the region, the Mountain Lion (*Puma concolor*), would be less likely to select a large male Bighorn Sheep when he was associating with smaller immature Aoudad. For Bighorn Sheep, the two most important antipredator strategies include group living and the proximity to abrupt escape terrain (Berger 1978; Shackleton 1985). Furthermore, during foraging by solitary and small groups (five or fewer) of Bighorn Sheep, interruptions to scan the environment are frequent and foraging efficiency is poor (Berger 1978).

Although Bighorn Sheep and Aoudad are known to compete for the same resources and habitat (Barrett 1967; Seegmiller and Simpson 1979; Simpson and Krysl 1981; Richardson 2007; Brewer and Hernádez 2011) and are noted to be behaviorally incompatible (Richardson 2007; Brewer and Hernádez 2011), the images from the camera-trap indicate that there is at least some passive tolerance among some of the individuals of the two species. Similarly, in Death Valley National Park, California, it generally was perceived that burros were causing great harm to native Bighorn Sheep due to competition between the two species (Welles and Welles 1961). Subsequent long-term research concluded that both species were abundant in the area, and that both used resources simultaneously and without strife (Welles and Welles 1961). Further research is warranted to ascertain the long-term impacts of Bighorn Sheep-Aoudad interactions at BBRSP.

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A Review and Synopsis of Nest Site Selection and Site Characteristics of Western Pond Turtles

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***Abstract*.—The Western Pond Turtle (*Actinemys marmorata*) is a widely occurring freshwater turtle in western North America where it occupies a variety of habitat types. Relatively little is known about its nesting ecology and nest site selection in these habitats and microhabitats. We compiled the known data on nest site characteristics along with new data from Moorhen Marsh, an 8.5-ha constructed wetland owned and operated by the Mt. View Sanitary District in Martinez, California. Because no standard set of characteristics has been used for measuring physical characteristics associated with nests sites of pond turtles, comparing data sets across study sites proved problematic. The two most reliable measurable characteristics were straight-line distance to water, and slope at or adjacent to nest sites. All reported nest sites (n = 505), when averaged, were approximately 51 m from the nearest water body, and the average slope (n = 193) was approximately 9%. Because availability of suitable nesting habitat is likely a limiting factor in some populations, we recommend that future studies use a standard set of characteristics when conducting research on *A. marmorata* nests.**

Key Words.—Actinemys marmorata; buffer; freshwater turtle; nesting behavior; plasticity; suitable

Introduction

The Western Pond Turtle (*Actinemys marmorata*) is an emydid turtle that occurs primarily west of the Sierra-Cascade ranges from the Puget Sound area in Washington State, south through Oregon and California, and into Baja California Norte (Stebbins 1985; Jennings and Hayes 1994; Thompson et al. 2016). *Actinemys marmorata* was originally described as two subspecies, *A. m. marmorata* and *A. m. pallida*, by Seelinger (1945). Recently, genetic studies suggested 2–3 species: the Northwestern Pond Turtle (*A. marmorata*), the Southwestern Pond Turtle (*A. pallida*), and one unnamed species (Spinks and Shaffer 2005; Spinks et al. 2014). Currently, regulatory agencies in California regard the turtle as a single species with two subspecies and we use this older convention and identify this turtle as *A. marmorata* in our paper.

Actinemys marmorata typically inhabits a variety of aquatic habitats, which it uses for foraging, refuge, dispersal, and winter torpor (Storer 1930; Spinks et al. 2003; Lechner 2004; Alvarez 2006; Germano 2010). The species is commonly associated with slow-moving streams, lakes, ponds, freshwater and slightly brackish wetlands, and human-made habitats, including treated wastewater effluent ponds, sewage treatment ponds, livestock ponds, and irrigation canals (Lechner 2004; Germano 2010; Bury et al. 2012b; Alvarez et al. 2014). *Actinemys marmorata* is also heavily dependent on upland areas near aquatic sites, using these for estivation, dispersal, and nesting (Rathbun et al. 2002; Spinks et al. 2003; Zaragoza et al. 2015).

Many populations of *A. marmorata* are declining (Thompson et al. 2016). This species is listed as Endangered in Washington State, as Sensitive-Critical in Oregon, and as a Species of Special Concern in California (Hays et al. 1999; Rosenberg et al. 2009; Bury et al. 2012a; Thompson et al. 2016). While there has been substantial research in recent years on the ecology, biology, and behavior of *A. marmorata*, significant gaps in knowledge, particularly the factors of upland ecology and its relationship to effective conservation and management, still exist (Rathbun et al. 2002; Lucas 2007; Scott et al. 2008; Rosenberg and Swift 2013; U.S. Fish and Wildlife Service 2015). Except for some general characteristics of nest sites in Ashton et al. (2012), synopses on the natural history of the species (Ernst and Lovich 2009; Bury and Germano 2008; Ashton et al. 2012) did not include information on the microhabitat characteristics of nest sites.

When compared with published research on other North American turtles (Lovich and Ennen 2013), investigations that focus on the nesting ecology of *A. marmorata* are limited (see: Rathbun et al. 1992; Rienche et al. 2019). *Actinemys marmorata* likely exhibit a recognizable pattern of nest site selection in terms of habitat type, vegetative cover, soil type, proximity to water, and seasonal timing of nest construction. Successful nest construction appears to be related to a number of factors: soil type; aspect of upland slope; distance from water and the associated flood plain; vegetation type and structure (density and height); and distance from habitat ecotones and human-made edges such as fences (Temple 1987; Rathbun et al. 1992; Spinks et al. 2003; Alvarez et al. 2014). St. John (2015) showed that 95% of her study group of the same species typically nested within 14.2 m of the nearest tree cover. Nest site fidelity has been documented in some female *A. marmorata*, but it is unclear just how widespread the phenomenon is in this species across its range (Crump 2001; St. John 2015). Here we review and analyze the existing available data

Figure 1. The 8.5 ha Moorhen Marsh, managed by the Mt. View Sanitary District (MVSD), showing six freshwater ponds and associated levees, Martinez, California, 2020. (Image taken from Google Earth 2020).

on nest site selection and the physical characteristics of nest sites of *A. marmorata* in light of data we collected.

Materials and Methods

We collected data from 2013 through 2015 on nest site characteristics of 68 nests of *A. marmorata* at Moorhen Marsh, 2.7 km east of the city of Martinez, California. Moorhen Marsh was constructed in1974 to secondarily treat effluent from the Mt. View Sanitary District (MVSD). Within the 9.5 ha freshwater wetland, six freshwater ponds of various sizes are separated by levees that also function as hiking trails (Fig. 1). The site is bordered by Interstate 680, the Shell Martinez Refinery, and the MVSD water treatment facility (Alvarez et al. 2014).

Actinemys marmorata use Moorhen Marsh for all aspects of their natural history. From 2013 to 2015, we used visual encounter surveys to closely monitor *A. marmorata* during the breeding season. Further, we attached nine UHF (MP2 units; AVM Instruments, Auburn, California, USA) and eight VHF (Quantum 4000E units; Telemetry Solutions, Concord, California, USA) radio transmitters to 17 turtles to facilitate finding nest sites. We marked nest sites and measured their characteristics: general soil type, slope percentage, slope aspect, distance to water, and any unusual conditions (i.e., proximity to trees, shrubs, fences, or anthropogenic structures). We also collected data on signs of predation, timing of nesting events, and nest abandonment.

In 2013, the detection of nest sites was facilitated by abundant sign of predation. During the 2014 reproductive season, we developed a monitoring protocol for nesting turtles that we used from 9 June through 14 July. Based primarily on the areas where turtles were known to nest in 2013, we conducted walking transects to monitor the project site from 1600 to approximately 2030 on most nights during a six-week period. If we found a female *A. marmorata* nesting or traveling overland, we closely monitored her and we recorded her behaviors while she completed the excavation and egg-laying process. Once nesting was complete, we hand-captured turtles as they departed, checked for existing identification marks (i.e., marginal scute notches), and marked new individuals if no identification mark was found. We also recorded weight, carapace length, and age, if annuli were present. Each female was immediately returned to the pond nearest to the location of capture. We flagged and caged each nest site for protection from predation (following Graham 1997) and recorded physical data from the site.

Results

We found 68 *A. marmorata* nests in 3 y: 13 in 2013, 32 in 2014, and 23 in 2015. Of the 13 nests we found in 2013, all were predated (we did not cage any nests). In 2014, all but two nests that we caged were predated (94% loss), and in 2015, we caged nine nests and 12 nests were predated (52% loss). We witnessed several females attempt to nest and leave before completing the nest, as well as females completing a nest but laying no eggs (Alvarez and Davidson 2018).

The 68 nests we recorded were an average of 9.4 m (range, 0.5–37 m) from the nearest wet edge of any pond

(Appendix). Additionally, we found that nests were located on slopes with an eastern or northern aspect, and a mean incline of 11.7% (0.0–32.8%). We found 30 (43.5 %) nest sites at Moorhen Marsh on level ground (i.e., no slope). Soil type was typically either hard-packed clay or silt-dominated but was cohesive and tightly compressed (Appendix). We found 41 (59.4%) nests located within 1-m of the perimeter fence.

Discussion

Our review of published literature on the nesting ecology of *A. marmorata* yielded just five relevant documents: two peer-reviewed journal articles and three Master's theses. Each reviewed document reported various types of nest site data, but these data varied considerably. These studies were done before established standards for measuring nest characteristics were published (Bury et al. 2012c: Pp. 118–119). To further the understanding of nest sites of *A. marmorata, we tabulated* reported characteristics and physical conditions for comparison (Table 1).

Understanding the components that make up quality habitat, and its availability of that habitat for nesting pond turtles, is critical to supporting the reproductive success of *A. marmorata.* Specific site characteristics such as ambient temperature, vegetation structure, soil type and compaction, slope and direction, distance from water sources, and placement relative to the flood plain likely affect clutch development, sex determination (and potentially population sex ratios), and overall reproductive output (Holte 1998; Lucas 2007; Gordon 2009; Christie and Giest 2017). Although there are published standards for reporting nest site characteristics for *A. marmorata* (Ashton et al. 2012), critical data are not always recorded, and data sets from one study site are often not comparable with another. Among all nests reported at Moorhen Marsh and in the literature combined (n = 505), the average distance to water was 51 m. Slopes associated with nest sites at Moorhen Marsh generally had either an eastern or northern aspect, but this may have been a limitation of the slope-orientation availability of the site. In contrast, at study sites other than Moorhen Marsh, nests were most frequently reported to face southern or southwestern directions. The average slope collected from the aggregate data (n = 193) from all sites was 9% (range, 0–60%), which is less than that of the average slope used at Moorhen Marsh (11.7%) alone. We did not measure the extent of slope availability, however, for all studies we reviewed. It is not known whether other reported sites offered a fuller range of slope orientations from which the turtles chose to nest facing south, southwest, or other. A wide range of slope availability, with a corresponding variety of vegetation composition would be required to adequately test slope aspect selection by *A. marmorata.* It does appear that this species avoids extensive shade at the Moorhen Marsh site (pers. comm.) but does nest in association with understory and overstory vegetation in Lake County, California (Bettelheim et al. 2006; St. John 2015). Further, pond turtles may use shaded areas for upland overwintering (Zargosa et al. 2015), suggesting that overstory vegetation plays a role in upland habitat use during some portions of the year.

Overall, the Moorhen Marsh study site includes inherent biases, in that slope availability is very limited; the site is situated on predominantly level or near-level ground with only west-, east-, and north-facing aspects. Further, the extent (i.e., distance from water) of uplands on the site is limited by a security fence, which may act as a barrier to some turtles seeking upland habitat sites for nesting (Alvarez et al. 2014). Notwithstanding these drawbacks, with the exception of 2011, we visually encountered at least one *A. marmorata* post-emergent turtle in Moorhen Marsh every year between 2008 and 2014. This suggests that, despite the limited upland conditions, *A. marmorata* will select from available habitat and microhabitat to produce successful nests, even when conditions do not include characteristics that may be considered optimal.

Table 1. Summarized data from Moorhen Marsh, and a review of data from various studies on nest site characteristics of *A. marmorata.* The abbreviation ADW = average distance to water, AS = average slope (% incline), NZS = number of nests with 0% slope, AVH = average vegetation height, and ND = no data.

Source	n	ADW (m)	AS	NZS	Aspect	Soil	AVH	Location
Moorhen Marsh	68	9.4	11.5	15	N, E, S	silt, clay	ND	Contra Costa County, California
Crump 2001	3	40	ND	ND	ND	sand, silt, clay	ND	Waddell Creek, Santa Cruz County, California
Rathbun et al. 2002	14	26.6	10.7	ND	ND	ND	ND	San Mateo County, California
Lucas 2008	23	56	10	2	SE	51% clay/silt	38.7 cm	Columbia River, Washington
Lucas 2008	12	33	10	1	NE	8% clay/silt	36.3 cm	Puget Sound, Washington
Bettelheim et al. 2006	24	3-15	ND	ND	ND	ND	ND	Clear Lake, California
Holte 1988	54–31	132.9	4.3	ND	ND	ND	ND	South Applegate, Fern Ridge Reservoir, Oregon
Holte 1988	12–8	48.2	3.7	ND	ND	ND	ND	Tripass, Fern Ridge Reservoir, Oregon
Holte 1988	16–9	171.1	1.8	5	ND	ND	ND	North Applegate, Fern Ridge Reservoir, Oregon
Holte 1988	27–12	5.6	12.4	ND	ND	ND	ND	South Marsh, Fern Ridge Reservoir, Oregon
Holte 1988	27–18	5.3	12.4	ND	ND	ND	ND	Kirk, Fern Ridge Reservoir, Oregon

The extensive range of *A. marmorata* suggests strong ecological plasticity in habitat use (Stebbins and McGinnis 2013). The species occurs in high and low deserts, grasslands, wetlands, riparian areas, and coniferous forests with moderate-gradient streams (Jennings and Hayes 1994; Bury and Germano 2008; Bury et al. 2012a), which suggests a correspondingly high level of plasticity in associated nest site selection. The common nest site selection factors, as currently understood from available literature and our fieldwork, appear to include adjacency to (i.e., within approximately 51 m of) aquatic refuge and feeding habitat. Factors influencing nest site selection, however, are undoubtedly influenced by the availability of appropriate habitat. Preferred habitat characteristics appear to include areas of sparse vegetation and significant solar exposure (Holte 1998; Rathbun et al. 2002; Bettelheim et al. 2006; Lucas 2007). Although soils at the Moorhen Marsh site are typically made up of cohesive silt and clay, some researchers have reported instances of sandy substrate being used for nesting (Storer 1930; Crump 2001). If appropriate conditions are in close proximity to aquatic refuge and feeding habitat, *A. marmorata* may create successful nests within 1–2 m of the edge of the water (pers. obs.). When necessary, however, this species may travel distances of 200 m or more to find suitable nesting microhabitat (Storer 1930; Rathbun et al. 2002). This plasticity in nesting habitat selection creates challenges for researchers and land managers but this plasticity provides critical flexibility for the species as it faces increasing threats from stochastic events and habitat loss.

Until recently, resource managers and conservationists have focused primarily on aquatic habitat for *A. marmorata*, but understanding the adaptability to, preferences for, and limitations in nest site selection of the species will be critical to conservation efforts. Proposed protection of habitat buffers should include a variety of physical characteristics surrounding aquatic features. Although the species appears to exhibit a high level of plasticity in nest site selection, if protection and management of *A. marmorata* is limited to compressed (i.e., narrow) upland areas the species may be excluded from appropriate nesting habitat, a condition that may go undetected in a population for many years or even decades (Holte 1998; Hays et al. 1999; Lucas 2007). Additionally, compressed uplands may concentrate predation pressures that can greatly reduce nesting success (Spinks et al. 2003; Alvarez et al. 2014). Although our study site suggests that compressed areas (< 10 m wide) may support *A. marmorata* for all aspects of its natural history, our findings do not imply that a 10-m wide upland protection zone is suitable for *A. marmorata*. We simply suggest that such an area, if it includes access to existing aquatic habitat, can be suitable for population sustainability.

Our study site is not typical or representative habitat. Ashton et al. (2012) recommends protection of a 50-m wide buffer area around a given aquatic feature. We feel that this will likely protect many nests from disturbance, however, we suggest that much work needs to be done to more accurately understand the extent of uplands that are utilized for nesting. We further suggest nest-surveys not be used to determine the presence of nests. The cryptic nature of pond turtle nests makes them extremely difficult to locate, even for highly skilled biologists. Until more thorough, and consistently comparable research can be conducted, we recommend that all upland areas, irrespective of slope aspect, slope incline, soil type, vegetation type, etc., be protected if it lay within 50 m of occupied or presumed occupied aquatic habitat.

Future research must include long-term investigations into a wider range of habitats and microhabitats used by the species for nesting, estivation, over-wintering, upland refuge, and upland dispersal routes (see: Semlitsch and Bodie 2003). There is also an urgent need for additional, ongoing efforts to better understand the factors surrounding nest site selection within various habitat types. A significant first step towards such efforts would be the use of standardized set of measurable nest site characteristics, along with an accepted protocol and associated data collection forms, so that data can be comparable across study areas throughout the species range. With the use of standardized data collection, stakeholders will have the ability to more accurately estimate nest site selection characteristics and could greatly enhance management of sites used for nesting by *A. marmorata.*

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Jeff A. Alvarez is a Herpetologist who has specialized in California reptiles and amphibians for 30 y. He has worked with California Red-legged Frogs (*Rana draytonii*), California Tiger Salamanders (*Ambystoma californiense*), Western Pond Turtles, and Alameda Whipsnakes (*Masticophis lateralis euyxanthus*) for much of his career. His focus is on the conservation and natural history of herpetofauna, survey techniques development, and conducting peer-level special-status species workshops. (Photographed by Kelly Davidson).

Appendix. Environmental data collected on 68 Western Pond Turtle (*Actinemys marmorata*) nests from Moorhen Marsh in Martinez, California, from 2013 through 2015. Data were collected from predated (P) nests and intact (I) nests. Slope aspect are N = north, E = east, S = south, and NW = northwest. General soil character was a subjective characteristic based on the surface structure in the immediate area of the located nest.

Date	Slope (%)	Slope aspect	Distance to water (m)	Predated/ Intact	General Soil Character		
					Friable	Hard clay	Rocky/gravel
25 May 2013	0.0	—	2.8	P		X	
5 June 2013	0.0	—	24.0	P		X	
6 June 2013	1.6	N	32.0	P		X	
8 June 2013	2.3	N	33.0	P		X	
13 June 2013	28.0	E	49.0	P	X		
16 June 2013	27.0	E	33.0	P	X		
18 June 2013	29.5	E	36.5	P	X		
21 June 2013	19.5	E	33.0	P	X		
22 June 2013	12.5	E	31.5	P	X		
22 June 2013	12.5	E	31.0	P	X		
24 June 2013	12.0	E	33.0	P	X		
27 June 2013	28.5	E	36.0	P	X		
1 July 2013	0.0	—	2.1	P	X		
5 June 2014	21.9	E	35.5	P	X		
5 June 2014	18.0	E	32.5	P	X		
5 June 2014	0.0	—	3.0	P		X	
5 June 2014	0.0	—	2.3	P		X	
5 June 2014	27.7	E	33.5	P	X		
5 June 2014	29.5	E	36.5	P	X		
5 June 2014	0.0	—	3.0	P	X		
5 June 2014	0.0	—	2.0	P	X		
7 June 2014	0.0	—	19.5	P		X	
7 June 2014	7.0	N	26.5	P		X	
7 June 2014	2.3	N	33.0	P		X	
7 June 2014	20.0	E	37.5	P	X		
7 June 2014	0.0	—	17.5	P		X	
7 June 2014	18.4	N	29.4	P		X	
7 June 2014	0.0	—	33.2	P		X	
7 June 2014	6.7	E	36.0	P	X		
10 June 2014	19.0	E	35.4	P	X		
10 June 2014	3.9	E	35.8	P	X		
13 June 2014	0.0	—	2.0	P		X	
14 June 2014	2.2	N	68.0	P		X	
14 June 2014	1.8	N	65.0	P		X	
17 June 2014	13.1	E	36.7	P	X		
23 June 2014	25.6	E	38.9	P	X		
23 June 2014	10.4	E	39.6	P	X		

Appendix (continued). Environmental data collected on 68 Western Pond Turtle (*Actinemys marmorata*) nests from Moorhen Marsh in Martinez, California, from 2013 through 2015. Data were collected from predated (P) nests and intact (I) nests. Slope aspect are N = north, E = east, S = south, and NW = northwest. General soil character was a subjective characteristic based on the surface structure in the immediate area of the located nest.

Date	Slope (%)	Slope aspect	Distance to water (m)	Predated/ Intact	General Soil Character		
					Friable	Hard clay	Rocky/gravel
23 June 2014	0	E	9.4	I	X		
28 June 2014	15.9	E	36.0	P			
30 June 2014	0.0	—	66.2	P	X		
30 June 2014	19.2	N	30.3	P	X		
1 July 2014	1.0	S	23.5	P			
1 July 2014	0.0	—	9.5	P	X		
3 July 2014	22.3	E	32.0	I	X		
9 July 2014	14.1	E	29.0	P	X		
27 May 2015	15.6	E	40.8	P			X
27 May 2015	32.8	E	33.5	P			X
27 May 2015	30.1	E	33.5	P			X
27 May 2015	28.2	E	31.8	P			X
28 May 2015	0.0	NW	26.0	I	X		
29 May 2015	6.1	N	64.0	P		X	X
29 May 2015	5.5	N	31.0	P		X	X
1 June 2015	16.1	E	35.4	I	X		
1 June 2015	10.4	N	27.8	P		X	X
2 June 2015	17.0	N	64.0	P	X		
3 June 2015	0.0	NW	16.5	I	X		
7 June 2015	7.3	E	39.8	I			X
9 June 2015	20.6	E	37.0	I			X
9 June 2015	7.4	—	3.0	I	X		X
10 June 2015	15.6	E	40.9	P			X
13 June 2015	16.7	E	35.6	P		X	
16 June 2015	13.9	E	32.5	I	X		
20 June 2015	4.4	NW	5.9	I	X	X	
22 June 2015	18.5	E	36.4	I		X	
25 June 2015	3.2	—	22.0	P	—	—	—
1 July 2015	24.2	N	30.4	I		X	
5 July 2015	7.3	E	37.4	P			X
7 July 2015	—	E	33.0	P			X

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NOTES

DISTRIBUTIONAL STATUS OF THE COMMON BLACK HAWK (*BUTEOGALLUS ANTHRACINUS*) IN CHIHUAHUA, MEXICO

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***Abstract.*—Updating species ranges and documenting range extensions with new municipality records helps monitor the status of populations, provides insight into life-history traits, and informs conservation decisions. Herein we document a new locality record and range extension of the Common Black Hawk (*Buteogallus anthracinus*) in Chihuahua, Mexico, which partially fills a distributional gap in the central part of the state. We also update information on the winter, migration, summer, and year-round distribution ranges of this species in Chihuahua. We compiled several sources of distributional information and produced new range maps based on the aerographic method. We show that the distribution of this species in Chihuahua is not only associated with wet drainages within Sierra Madre Occidental Montane Forests and Sinaloan Dry Forests ecoregions as previously thought, but that the species is also a regular summer resident in riparian habitats along the Chihuahuan Desert, and an accidental migrant in that ecoregion.**

Key Words.—aerographic method; tropical hawk; range extension; riparian areas

The Common Black Hawk, *Buteogallus anthracinus*, is a buteonine raptor that depends on riparian areas with a range extending from the southwestern U.S. to northern South America (Schnell, J.H. 2020. Common Black Hawk (*Buteogallus anthracinus*), version 1.0. *in* Birds of the World. Poole, A.F. and F.B. Gill (Eds.). Cornell Lab of Ornithology, Ithaca, New York. Available from https://doi.org/10.2173/bow.comblh1.01 [Accessed 5 June 2020]). It typically nests in trees but occasionally nests on cliffs (Schnell *op. cit*.). Common Black-Hawks are obligated to wet riparian habitats where they often hunt fish, amphibians, and reptiles from perches just above the water or while walking along shore (Russell and Monson 1998; Flesch 2008a). The species has expanded its summer range in northern Mexico and southwestern U.S. (Schnell *op. cit*.). Its breeding range in the U.S. is limited to the southwestern corner in Utah and northwestern portion in Arizona along streams of Virgin River drainage (Wauer and Russell 1967; Gifford 1985). Small breeding populations exist in Texas, where it is a rare and local summer resident in the Davis Mountains and adjacent localities (Peterson and Zimmer 1998), and a regular nester along the Rio Grande in southern Brewster and Presidio Counties, and in the Concho Valley (Lockwood and Freeman 2014).

In Mexico, the Common Black Hawk has been recorded as a common to fairly common resident of southern Sonora and eastern Nuevo Leon, and southern Tamaulipas to the Yucatan Peninsula (Howell and Webb 1995). Although the summer range of the Common Black Hawk in Chihuahua and Sonora has been mapped to the Sierra Madre Occidental ecoregion (Howell and Webb 1995), there are records wholly within the Sonoran Desert ecoregion just west of the Sky Islands in southeastern Arizona and northern Mexico (Russell and Monson 1998; Flesch 2008b). To our knowledge, there have been no concerted efforts to determine the distribution of Common Black Hawks across Chihuahua, and very little research of any kind has been published on this species in this part of its northern distributional range. Herein we document a new locality record and range extension in Chihuahua, which partially fills a distributional gap in the central part of the state. We also update the winter, migration, summer, and year-round distribution ranges of Common Black Hawk in Chihuahua, Mexico.

We compiled Chihuahua records of the Common Black Hawk from thee major resources: published literature (Stager 1954; Vuilleumier and Williams 1964; Gómez de Silva 2005, 2008; Moreno-Contreras et al. 2015), the Global Biodiversity Information Facility (GBIF; www.gbif.org [Accessed 8 June 2020]), and specimen records in the Atlas of Mexican Bird Distributions (Peterson et al. 2016), a database that has focused on comprehensive capture of distributional data from Mexican bird specimens housed in 70 scientific collections. The GBIF data was downloaded using the occ_search function of rgbif R-library (Chamberlain et al. 2020). These distributional data include records from Naturalista, an online social network of people sharing biodiversity information to help each other learn about nature; (https://www.naturalista.mx [Accessed 8 June 2020]), and eBird, a real-time checklist program and online citizen-based platform for collection of ornithological data (https://ebird.org [Accessed 8 June 2020]).

As an additional automated step, we employed the clean_coordinates function of the CoordinateCleaner

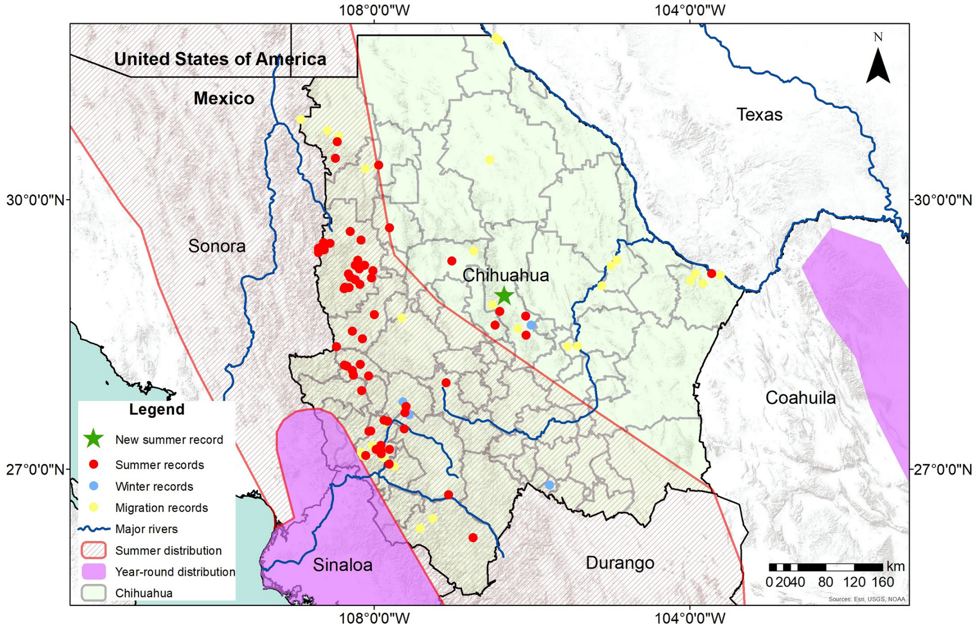

Figure 1. Digitized representations of published range maps of the Common Black Hawk (*Buteogallus anthracinus*) in Chihuahua of Howell and Webb (1995) supplemented by BirdLife International and Handbook of the Birds of the World (http://datazone.birdlife.org). All records separated by seasonal status (including a new record at Cueva de las Monas, Chihuahua Municipality) and were overlaid on the published distribution maps.

R-package (Zizka et al. 2019) as a bioinformatic pipeline filter of the GBIF occurrence data to avoid sampling bias in georeferenced data (i.e., doubtful records based on known distributional ranges). As a manual step, we individually checked the Common Black Hawk observations in the state of Chihuahua and changed the geographical coordinates of an occurrence record of eBird indicating Chihuahua City, but which actually referred to Mil Castillos. We only included records submitted to Naturalista if they were classified as research-grade, which requires a photograph and corroborated identification by at least one other user. We did not exclude potentially erroneous observations from the other data sources, largely because necessary details regarding the observations were rarely available to us. We conducted the cleaning process for Common Black observations using R 3.6.1 (R Development Core Team 2019).

Once we gathered all the available distributional information and after removing duplicates (records submitted to multiple databases), we plotted the locations of all records of Common Black Hawk occurrence by seasonal status to map their spatiotemporal distribution. We used the contemporary distribution range maps of Howell and Webb (1995) to determine what are considered out-of-range records. These maps were modified and supplimented by records from BirdLife International and the Handbook of the Birds of the World (http://datazone.birdlife.org [Accessed 4 June 2020]).

We considered three seasonal categories based on published research of the life history of Common Black Hawks (Russell and Monson 1998; Flesch 2008a; Schnell *op. cit.*): winter (November to early February), migration (spring: mid-February to July; fall: late August to October), and summer (April to late September in breeding habitat). We determined the habitat type of Common Black Hawk observations by overlaying them with the land use and vegetation cartography (Scale 1:50,000) of the state of Chihuahua provided by CONAFOR (https://www.cnf.gob.mx:8443/snif/seif_chihuahua/). We constructed the new distributional maps for three periods (winter, migration, and summer) employing the aerographic method (Rapoport 1982). This method uses geo-referenced locality data. Occurrence points are interconnected to form an open, minimum spanning tree (MST), where all points are connected by their shortest distance. The minimum distances between pairs of points are measured and the standard deviation distance (SD propinquity index) is calculated (Rapoport 1982). We created the MST in PASSaGE 2 (Rosenberg and Anderson 2011), where we considered the minimum distance (SD propinquity index) as the radius of each locality point. The cumulative area of the circles (deducting overlapping fragments) is taken as the species distribution area. To assess potential changes in distribution within Chihuahuan protected areas, we compared through a gap analysis (Moreno-Contreras et al. 2017) the breeding distribution across two time periods: historical (1934–2000) and current (1934–2019) based on the timing of records. We performed all geospatial work using ArcMap 10.3.1 (Esri, Redlands, California, USA).

We found 156 records of Common Black Hawks for Chihuahua in the compiled database from 1934 to 2020 (Fig. 1); of these, four records are based on specimens from Chihuahua from three localities along the Sinaloan

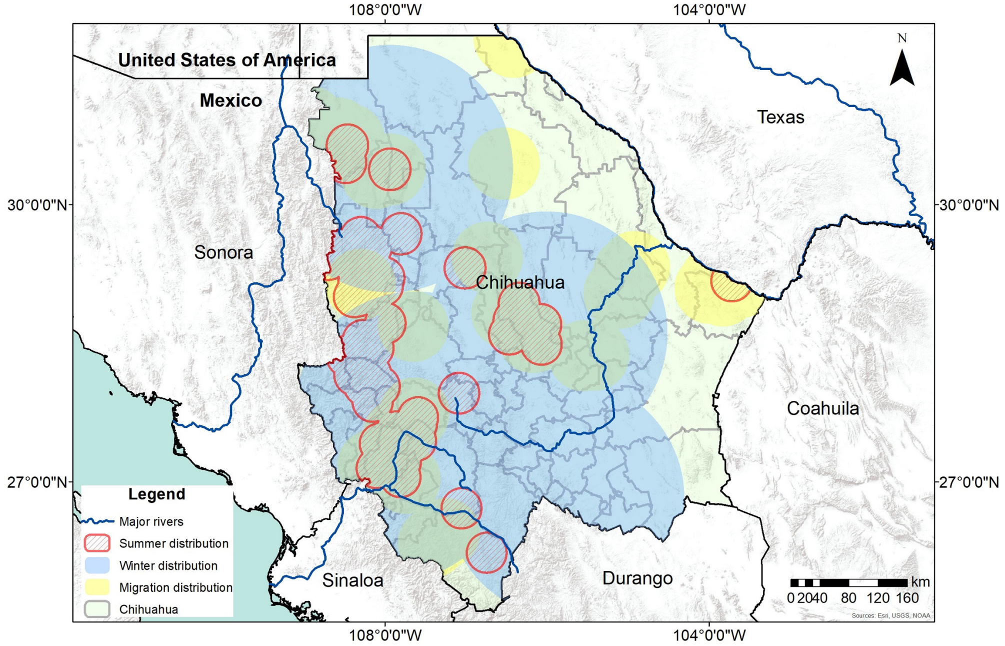

Figure 2. New distribution maps based on the aerographic method representing the occurrence of the Common Black Hawk (*Buteogallus anthracinus*) in Chihuahua, Mexico.

Dry Forest ecoregion from 1934 to 1950. All specimens are from the Barrancas del Cobre region (also named as Cooper Canyon), where historically the species has been mapped as a year-round resident in the Sinaloan Dry Forest ecoregion (Howell and Webb 1995). In that region, R. T. Moore collected one male at camp # 1 near La Mesita on 28 May 1934 (ID 491; Moore Laboratory of Zoology [MLZ], Los Angeles, California, USA). Stager (1954) mentioned that the Common Black Hawk appeared to be a rather abundant species in southwestern portions of this ecoregion and he collected at several elevations and in varying vegetation associations: Arroyo Hondo (adult male on 10 May 1950; ID 612, Natural History Museum of Los Angeles County [LACM], Los Angeles, California, USA) and along the Urique River in the bottom of the Barranca del Cobre (on 18 May 1950; ID 619, Louisiana State University Museum of Natural Science [LSUMZ], Baton Rouge, Louisiana, USA).

Our geospatial analyses revealed that in Chihuahua, the Common Black Hawk has a summer distribution range of 45,345 km^2 (Fig. 2), whereas the historical summer range of the species covered 52,958 km^2 (Fig. 3). The extension of Chihuahuan winter distribution of the Common Black Hawk was 189,351 km^2 and its

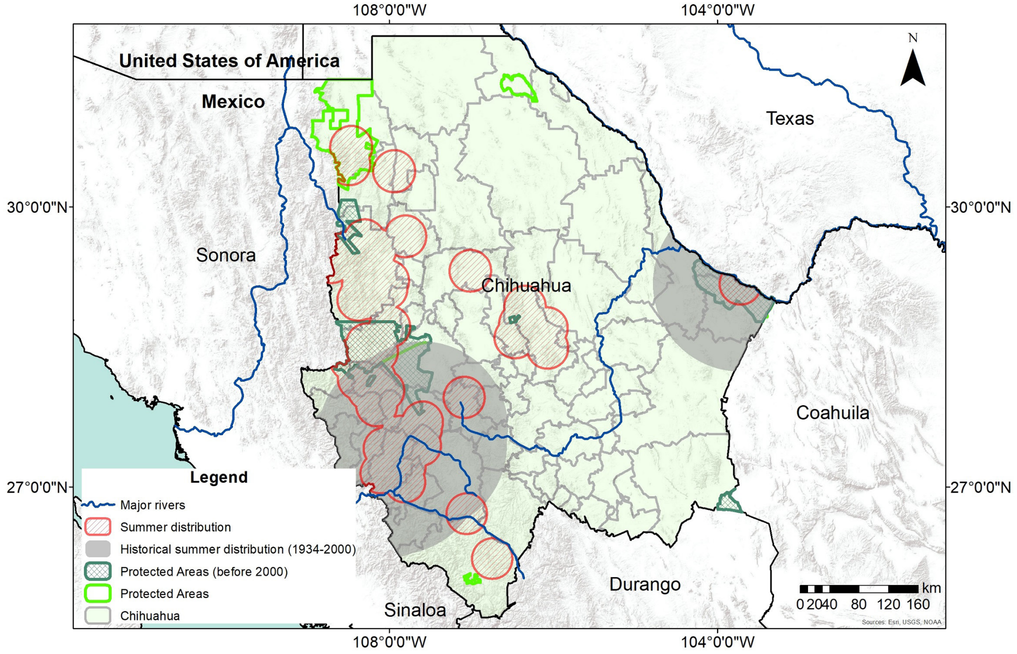

Figure 3. Historic and current summer ranges of Common Black Hawk (*Buteogallus anthracinus*) within Chihuahuan protected areas.

migration distribution covered 86,902 km^2 (Fig. 2). The summer range has remained underrepresented within the network of protected areas (Fig. 3) both historically (11% of protected range) and currently (16% of protected range) The summer distribution (both historical and current ones) includes protected areas such as Bassaseachic Falls, Santa Elena Canyon, Tutuaca, and Papigochic (Fig. 3). The species is considered to have a strong affinity to the wet riparian systems within Sierra Madre Occidental ecoregion in the western portion of Chihuahua (Howell and Webb 1995) during summer. There are at least five so called out-of-range localities, however, in the central part of Chihuahua and one locality in the eastern portion of the state (i.e., Santa Elena Canyon protected area) suggesting a regular local summer occurrence (Fig. 1 and 2).

On 28 May and 11 June 2020, the lead author, accompanied by other bird watchers during ornithological surveys in the Punta de Agua area, Ejido Cuauhtémoc, Chihuahua Municipality, observed a presumably breeding pair of Common Black Hawks at the Cueva de las Monas archaeological site. This area is a recharge area for a phreatic zone that supplies water to nearby Chihuahua City, which is quite close to two small populated areas, Punta de Agua and Ejido Cuauhtémoc (28°56'28.3"N, 106°20'36.2"W). The vegetation communities from the Punta de Agua area to the nest observation site consisted of a mix of typical desert scrub plants, such as Creosote Bush (*Larrea tridentata*), with a riparian zone and oak forest (*Quercus* spp.) of medium height. We noted that one Common Black Hawk vocalized while chasing away five Turkey Vultures (*Cathartes aura*) that were in the area. The first author found and photographed the pair of Common Black Hawks and their nest in a dead oak tree (*Quercus* spp.), always at a suitable distance to prevent the pair from leaving the nest. On 22 June 2020, a birdwatcher (Bonifacio López) recorded a Common Black Hawk on the nest previously observed by the first author (Fig. 4). This nesting record is about 52 km away from the known summer distribution of the Common Black Hawk in Chihuahua (Fig. 1).

Although the species is mostly considered a summer resident in the northern Sierra Madre Occidental and a year-round resident in the most southern Oak-Pine Forests (Fig. 5), there are also several winter records (Fig. 5). Single individuals have been reported at the town of Casas Grandes on 23 February 1984 (eBird), in an irrigation pond on the east edge of Casas Grandes Municipality on 5 January 2008 (Gómez de Silva 2008), in Chihuahua Municipality on 25 January 2019 (Naturalista), and in Hidalgo del Parral Municipality on 30 December 2018 (Naturalista). In Sonora, there are recent records of wintering birds within about 15 km of Chihuahua in the Aros River canyon. Because these records in Sonora are along the Aros River, which extends into western Chihuahua, it is probable that Common Black Hawks winter in this part of Chihuahua (Flesch et al. 2015).

Figure 4. Female Common Black Hawk (*Buteogallus anthracinus*) nesting at Cueva de las Monas, Chihuahua, Mexico, 22 June 2020. (Photographed by Bonifacio López).

Outside of its mapped distribution in montane forests and adjacent habitats in northern Mexico, Common Black Hawks likely are accidental transient in the Chihuahuan Desert ecoregion during the migration period. Records during this period include one at San Rafael, Ahumada Municipality 19 April 2008 (eBird) and in urban green spaces at Club Campestre 30 March 2015 (Moreno-Contreras et al. 2015) and at El Chamizal 18 March 2016 (observation by Bruno Lima and Karina Avila, http://www.wikiaves.com/2057166 [Accessed 15 June 2020]), both in Juarez Municipality. Most of the eastern records in Coyame del Sotol Municipality are migrant individuals associated with agricultural fields and microphyllous desert scrub (see Fig. 5) within the Chihuahuan Desert ecoregion (Vuilleumier and Williams 1964).

We have shown that the distribution of this species in Chihuahua is not only associated with wet drainages within Montane Forests of the Sierra Madre Occidental ecoregion and Dry Forests of the Sinaloan ecoregion as previously thought, but that the species is also a regular summer resident in riparian habitats along the Chihuahuan Desert, with scattered out-of-range winter records in northern (one January record), central (one

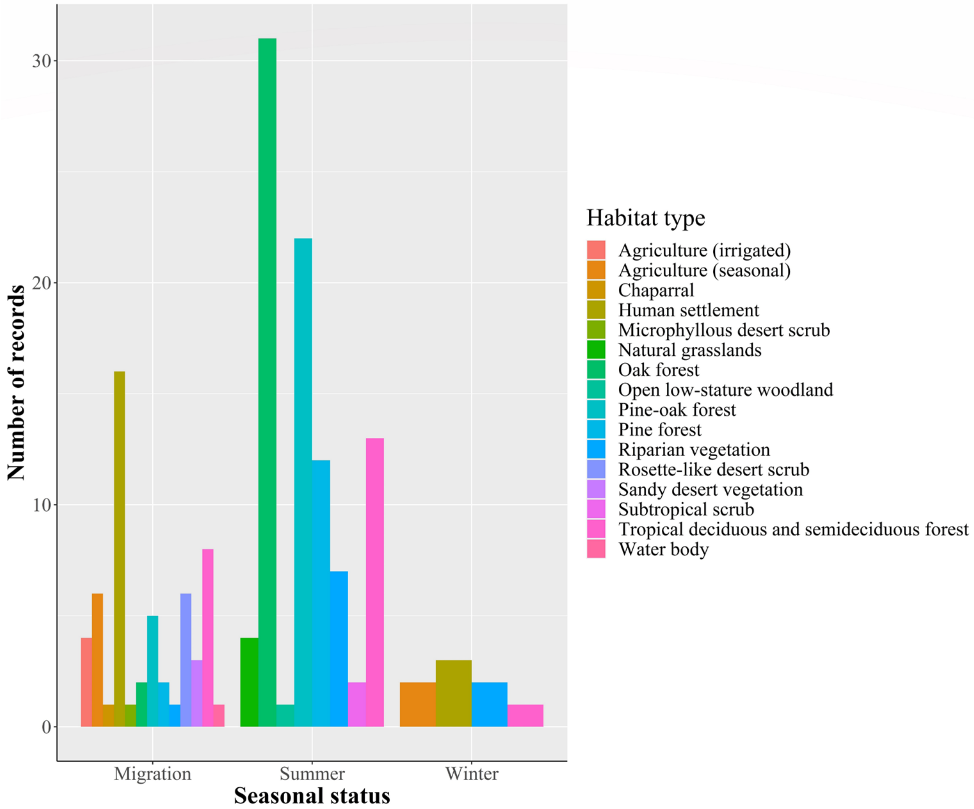

Figure 5. The number of records of the Common Black Hawk (*Buteogallus anthracinus*) by seasonal status and habitat type.

January record), and southern (one December record) portions of that state. There are four February records that are thought to be migrants or early arriving breeders, three of which were reported in human settlements along the Sierra Madre Occidental (e.g. Casas Grandes, Batopilas). In addition, Common Black Hawks tend to migrate in human-made habitats (i.e., urban green spaces of Juárez Muncipality) in the Chihuahuan Desert ecoregion (Moreno-Contreras et al. 2015). In fact, most of the records during migration come from human settlements (>10 records; Fig. 5). As reported for the state of Sonora, the species is probably attracted to the greater presence and permanence of water that supports breeding in Chihuahuan montane habitats compared to desert landscapes (Flesch et al. 2008b).

The Common Black Hawk is currently listed as Special Protection by Mexican law (NOM-059-SEMARNAT-2010). Although its breeding (e.g., riparian systems, pine-oak forests) and non-breeding habitats (e.g., microphyllous desert scrub) are relatively well represented within the network of protected areas of Chihuahua (Moreno-Contreras et al. 2017), much of its breeding habitat is greatly threatened by the clearing or alteration of riparian habitat (generally for agriculture), water diversion for irrigation and storage, diking and damming for flood control, lowering of the water table by pumping, and by livestock grazing, which eliminates regenerative seedlings (Schnell *op. cit*.). In addition, the historic and current range of this species is poorly represented in the Chihuahuan protected areas network (< 20% of protected distribution range). As anthropogenic activity continues to alter landscapes and change habitat suitability for this species and other organisms, documenting range shifts and monitoring population trends will be necessary for effective conservation assessment of this tropical hawk species.

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Israel Moreno-Contreras is currently a Ph.D. student at Universidad Nacional Autónoma de México, Mexico, in the Department of Evolutionary Biology. He completed his B.S. in Biology at Universidad Autónoma de Ciudad Juárez, Chihuahua, Mexico. Israel is interested in using genomics and ecological niche modeling to look at phylogeography, speciation, and conservation in Mesoamerican avifauna. (Photographed by Manuel Grosselet).

2020 ANNUAL MEETING REVIEW

Program Chair: **Kelly Holland,** GEI Consulting

Attendance: 601 participants including 134 students and 49 Early Career Professionals.

Plenary Theme: *Navigating the Intersections of Science and Policy*

Plenary Speakers: **Dr. Jennifer Norris,** Sacramento Fish and Wildlife Office; **Michael Peterson**, California Department of Insurance; **David Willms,** National Wildlife Federation; **Kim Delfino,** Defenders of Wildlife; **Kevin Hunting,** California Natural Resources Agency.

Western Section of The Wildlife Society
2020 Annual Meeting — Redding, California

Keynote Address: ***Wildlife Disease 2020 and Beyond: Moving from Microscopes to Policy***

Keynote Speakers: **Peregrine Wolff**, Wildlife Disease Association; **Deana Clifford**, CA Dept. Fish & Wildlife / UC Davis; **Dave Jessup,** UC Davis Wildlife Health Center

Awards Bestowed:

- The Raymond F. Dasmann Award for the Professional of the Year went to **Dr. Deana Clifford.**
- The Conservationist of the Year Award went to **Save the Redwoods League.**
- 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 **John McNerney.**
- The Barrett A. Garrison Mentor of the Year Award went to **Jeffrey Wilcox.**
- The TWS Fellows Award went to **Dr. Kelley Stewart.**
- The TWS Distinguished Service Award was awarded to **Dr. Reginald H. Barrett.**

Student Presentation Awards:

Oral Presentations

1st Place – Sophie Preckler-Quisquater, Does Natural Selection Affect Gene Flow Between Nonnative and Native Red Fox Populations? (UC Davis)

2nd Place – Anna Jordan, The Color of Conservation: Analyzing Color Polymorphism in the Giant Gartersnake (UC Davis)

3rd Place (tie)

Ryan Baumbusch, Diet Composition of Barred Owls (*Strix varia*) in the Pacific Northwest (Oregon State)

and **Anna Cassady,** Wastewater treatment plants and endangered & threatened wildlife species overlap in California watersheds (UC Riverside)

Poster Presentations

1st Place – Taylor Davis, Optimizing Tissue Sampling and Extraction Protocols for Next-Generation Genomic Sequencing (UC Davis)

2nd Place – Rachel Morrow, Use of DNA Sequencing to Identify the Origin of Northwestern and Southwestern Pond Turtles in Captive Breeding Programs (Fresno State)

3rd Place – Amanda Coen, Going beyond Broodstock Based Conservation: Genetic Diversity in White Sturgeon (*Acipenser transmontanus*) Repatriated Larvae in the Snake River, Idaho (UC Davis)

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Chapter Representatives

California Central Coast
Clint Scheuerman
Environmental Consultant

California North Coast
Elizabeth Meisman
GHD Inc.

Hawaii
Laura Luther
University of Hawai'i

Nevada
Mitchell Gritts
Nevada Dept of Wildlife

Sacramento-Shasta
Carlos Alvarado
Ascent Environmental

San Francisco Bay Area
Natasha Dvorak
Swaim Biological, Inc.

San Joaquin Valley
Randi McCormick
McCormick Biological

Southern California
Jeff Lincer

Committee Chairs

Awards and Grants
Richard Burg
CA Dept. Fish & Wildlife

Communications
Susanne Marczak
SD Zoo Institute for Conservation Research

Conservation Affairs
Kelly Holland
GEI Consultants

Diversity Committee
Bayan Ahmed
Dept of Water Resources

Membership/Historian
Don Yasuda
USDA Forest Service

Professional Development
Janine Payne

Western Wildlife Journal
Howard Clark
Colibri Ecological Consulting

Student Affairs
Katie Smith
WRA Environmental

Contractors

Bookkeeper
Mike Chapel

Program Director
Cynthia Perrine

Project Manager and Meeting Planner
Candace Renger

Webmaster
Eric Renger

Workshop Coordinator
Ivan Parr

2020 Western Section Membership

Regular:	547	Life-Partial:	5
Student:	166	Honorary:	1
New Professional:	92	Supporting:	5
Retired:	47	Contributing:	2
Life-Full:	37	**Total:**	**902**