

Coachella Valley Multiple Species Habitat Conservation Plan/ Natural Community Conservation Plan

2020 Annual Report



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We gratefully acknowledge Kristin Cummings whose beautiful artwork is our cover this year. Kristin submitted this poster as her capstone project for the California Naturalist program at UC Riverside - Palm Desert.

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I. Introduction

This Annual Report describes the progress made on implementation of the Coachella Valley Multiple Species Habitat Conservation Plan for the 2020 calendar year. Acquisition of key properties continued, with 2,325 acres added to the reserve system to protect habitat for our desert plants and animals. The CVCC acquired 2,075 of those acres. The map in Figure 2 on page 3 tells the story: we have made significant progress, with 98,369 acres acquired to date. At the same time authorized disturbance in conservation areas was at zero acres in 2020; since 1996, only 142 acres of authorized disturbance have occurred in conservation areas. Development outside the conservation areas, including important road projects and water infrastructure continued through the streamlined process facilitated by the Plan.

Biological monitoring activities continue to gather important data on covered species, including a genetic study of desert tortoise, ongoing monitoring of sand dune species, and a study about bighorn sheep and recreational trail use. The information gathered through the biological monitoring program helps us better manage our reserve lands and ensure the survival of the 27 plant and animal species the Plan is charged with protecting. On the land management side, we continue to install fences to protect lands from illegal dumping and vehicle trespass. Crews cleared invasive tamarisk from our properties at Willow Hole and North Shore Ranch. An effort to provide law enforcement capacity to address these concerns was initiated in 2020. And we cleaned up almost 3 tons of refuse from reserve lands. We appreciate the support of the members of the CVCC, our partners, and collaborators for the ongoing success of this visionary Plan.

Plan Background

The Coachella Valley Multiple Species Habitat Conservation Plan/Natural Community Conservation Plan (CVMSHCP) is a multi-agency conservation plan that provides for the long-term conservation of ecological diversity in the Coachella Valley region of southern California. The CVMSHCP includes an area of approximately 1.1 million acres and incorporates the watersheds within the jurisdictional boundaries of the Coachella Valley Association of Governments (CVAG). Tribal lands are not included in the CVMSHCP although coordination and collaboration with tribal governments has been ongoing. State and federal permits were issued in October 2008 and run for a 75 year term, during which the CVMSHCP is expected to be fully implemented and funded.

The Coachella Valley Conservation Commission (CVCC) was established in 2008 to oversee CVMSHCP implementation, and is comprised of elected officials from Riverside County, the cities of Cathedral City, Coachella, Desert Hot Springs, Indian Wells, Indio, La Quinta, Palm Desert, Palm Springs, and Rancho Mirage, as well as the Coachella Valley Water District, Mission Springs Water District, and the Imperial Irrigation District. The Riverside County Flood Control and Water Conservation District, Riverside County Regional Park and Open Space District, and Riverside County Waste Resources Management District are also members, as are the California Department of Parks and Recreation, the Coachella Valley Mountains Conservancy (CVMC), and the California Department of Transportation (Caltrans). Collectively, with the addition of CVAG, these entities constitute the CVMSHCP Permittees.

The CVMSHCP established a Reserve System to ensure the conservation of 27 Covered Species, 23 natural communities, and 3 essential ecological processes in perpetuity. This Reserve System consists of 21 priority Conservation Areas built around existing protected lands managed by local, state, or federal agencies and non-profit conservation organizations. To

complete the assembly of the Reserve System, lands are acquired or otherwise conserved (1) by the CVCC directly on behalf of the Permittees, (2) through state and federal agencies to meet their obligations under the CVMSHCP, or (3) through complementary conservation, whereby lands are acquired to consolidate public ownership in areas such as Joshua Tree National Park and the Santa Rosa and San Jacinto Mountains National Monument. Complementary conservation is not a Permittee obligation but does benefit the Plan.

In addition to acquisition, land in the Reserve System may be conserved through dedication, deed restriction, granting of a conservation easement, or other means of permanent conservation. To meet the goals of the CVMSHCP, the Permittees are obligated to acquire or otherwise conserve 100,600 acres in the Reserve System. State and federal agencies are expected to acquire 39,850 acres of conservation land. Complementary conservation is anticipated to add an additional 69,290 acres to the MSHCP Reserve System.

This Annual Report describes the activities for the calendar year from January 1, 2020 to December 31, 2020. As required by Section 6.4 of the CVMSHCP, this Annual Report will be presented at the CVCC meeting of April 8, 2021, which will serve as a public workshop. The report is also posted and available to the public on the CVMSHCP website, www.cvmshcp.org.

II. Reserve Assembly Progress

As of December 31st, 2020, Permittees have conserved 13,792 acres, almost 14% of their conservation goal (Figure 1). State and federal conservation has reached 23,954 acres, or 60% of their required contribution, and complementary conservation has accounted for 60,623 acres, about 87% of the anticipated acreage. Since 1996, 98,369 acres have been conserved under the CVMSHCP, with the assembly of the Reserve System about 47% complete (Table 1, Figure 2). A description of how CVCC allocates acreage credit is included in Appendix I.

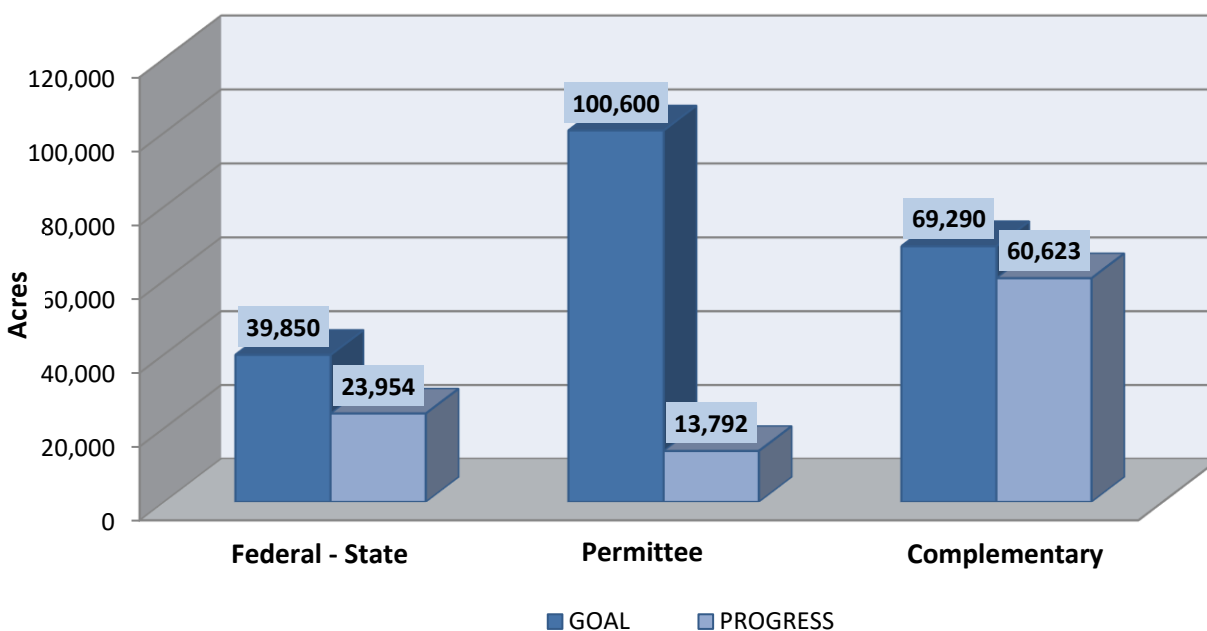


Figure 1: CVMSHCP progress toward conservation goals.

Land Acquisitions in 2020

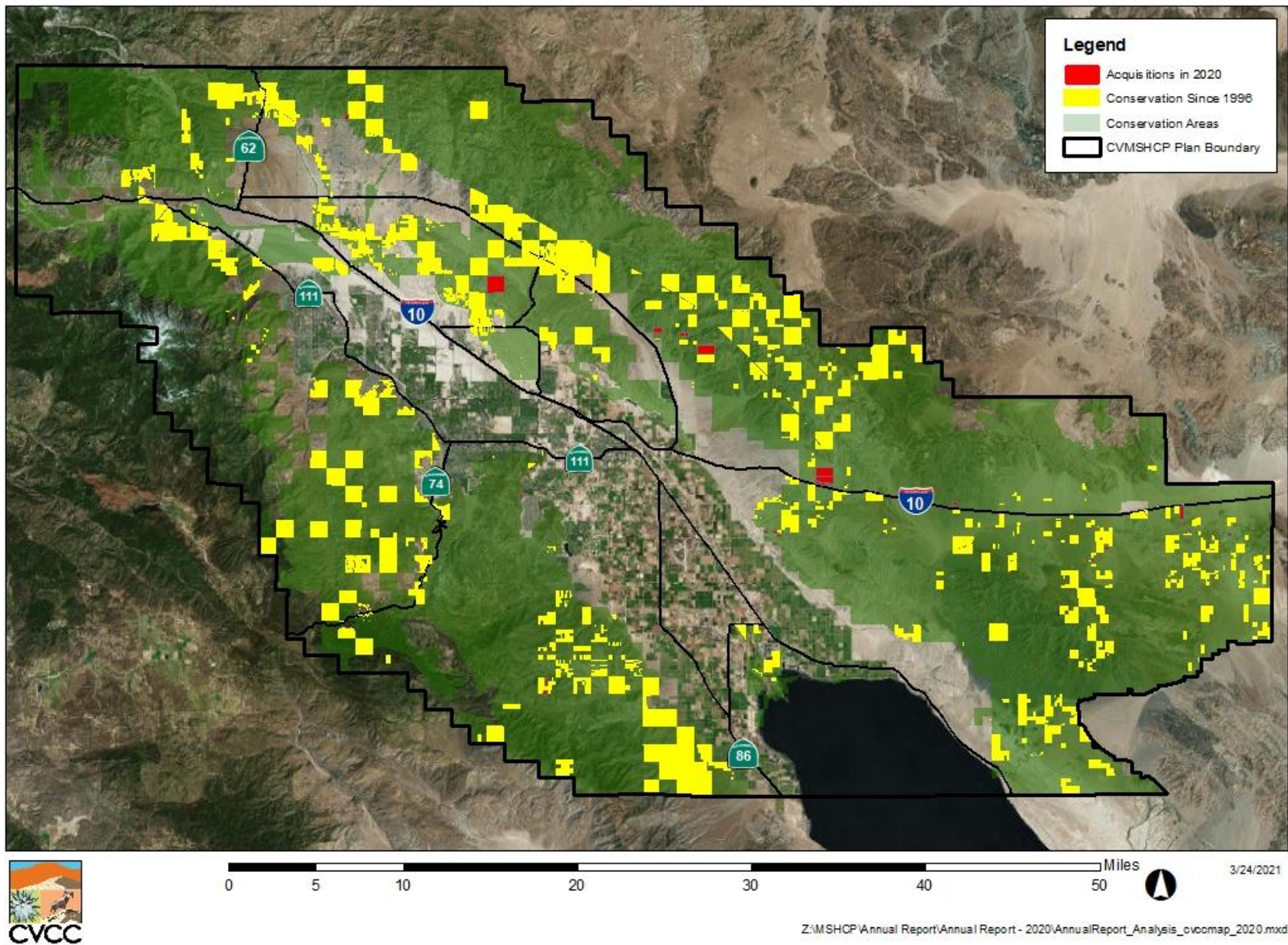


Figure 2: CVMSHCP Reserve assembly status, including 2020 acquisitions.

Once acquired, lands within the Reserve System are held in public or private ownership and are managed for habitat conservation and open space values. Land acquired under complementary conservation is often transferred in fee to either a state or federal agency or to a permittee for long term management (Table 2). Management of these lands contributes to the conservation of the Covered Species and the conserved natural communities included in the Plan.

Table 1: Summary of annual progress on reserve assembly.

Conservation Credit	Goal	Total Progress	1996 - 2008	2009-2015	2016	2017	2018	2019	2020
Federal - State	39,850	23,954	15,538	6,492	319	525	816	224	40
Permittee	100,600	13,792	5,567	3,589	799	793	578	391	2,075
Complementary	69,290	60,623	42,847	13,835	612	1,703	906	510	210
Total	209,740	98,369	63,952	23,916	1,730	3,021	2,300	1,125	2,325

Table 2: Acres of management credit.

Management Credit	Progress (acres)
Federal - State	60,632
Permittee	16,030
Complementary	21,707
Total	98,369

Land Acquisition to Achieve the Conservation Goals and Objectives

In 2020, CVCC completed 22 transactions acquiring 29 parcels totaling 2,075 acres at a cost of \$1,293,253 in CVCC funds (Table 3). Throughout 2020, local, state and federal partners acquired lands in six different Conservation Areas, with 65% of the acres acquired located in the Desert Tortoise and Linkage Conservation Area (Figure 3). Similarly, CVCC acquisitions on behalf of local Permittees occurred predominantly in the Desert Tortoise and Linkage Conservation Area, as well as four other Conservation Areas (Figure 4). All lands conserved by CVCC and partner organizations during the period from January 1, 2020 to December 31, 2020 are depicted in Figure 2 and listed in Appendix II.

CVCC acquires lands with funding from CVMSHCP development mitigation fees as well as CVAG and other agency contributions to mitigate for regional roads and other transportation projects. In addition, funding from land acquisition partners continues to be an important source of land acquisition dollars (Figure 5). Significant federal funding has been provided through the U.S. Fish and Wildlife Service's Cooperative Endangered Species Conservation Fund, referred to as Section 6. State funding comes from several sources. The Coachella Valley Mountains Conservancy contributes significantly to the acquisition of conservation lands through grants to various organizations, including CVCC. The Wildlife Conservation Board and California Department of Fish and Wildlife are both major source of State funding. The non-profit Friends of the Desert Mountains has acquired lands using grants from CVMC, private donations, and other sources; many of these lands have been transferred to CVCC. Other agencies and non-profits have also provided funds for land conservation. CVCC gratefully acknowledges the support from our partners.

Table 3: Lands acquired by CVCC in 2020.

Project	Conservation Area	Acres	Purchase Price
A199-UMC-K&R	Upper Mission Creek/Big Morongo Canyon	5.41	\$155,000
A199-UMC-K&R	Upper Mission Creek/Big Morongo Canyon	18.00	\$0
A200-UMC-Welty	Upper Mission Creek/Big Morongo Canyon	3.10	\$18,600
Barnes	Desert Tortoise and Linkage	47.09	\$21,365
Barnes	Desert Tortoise and Linkage	320.80	\$140,733
Barnes	Desert Tortoise and Linkage	232.25	\$101,169
Barnes	Desert Tortoise and Linkage	36.59	\$16,733
Buck	Desert Tortoise and Linkage	3.04	\$2,000
Buck	Desert Tortoise and Linkage	22.47	\$11,000
Buttram-McClean	Desert Tortoise and Linkage	10.02	\$4,500
Day Construction Corp	Thousand Palms	5.08	\$50,000
DeVito	Thousand Palms	2.37	\$13,000
Dillard	Desert Tortoise and Linkage	20.08	\$9,000
Gray	Willow Hole	4.60	\$14,000
Holland	Desert Tortoise and Linkage	40.12	\$18,000
Horowitz	Desert Tortoise and Linkage	40.13	\$18,000
Isaacs et al (Ghoreichi, Glesby, White, Adler, Levin)	Desert Tortoise and Linkage	99.69	\$45,000
Keck-Horn	Willow Hole	4.99	\$15,000
Lazar Family Trust	Thousand Palms	646.28	\$291,353
Lechuga	Desert Tortoise and Linkage	20.04	\$9,000
National Rec	Desert Tortoise and Linkage	115.36	\$60,000
Nevarez	Desert Tortoise and Linkage	19.97	\$9,000
Njau	Willow Hole	5.01	\$15,000
Panhwar	Willow Hole	5.05	\$7,500
Panhwar	Willow Hole	4.95	\$7,500
Ruppert	Desert Tortoise and Linkage	313.10	\$140,800
Shovlin et al	Upper Mission Creek/Big Morongo Canyon	15.59	\$45,000
Shovlin et al	Upper Mission Creek/Big Morongo Canyon	5.15	\$15,000
Steinberg	Thousand Palms	8.40	\$40,000
Total		2,074.73	\$1,293,253

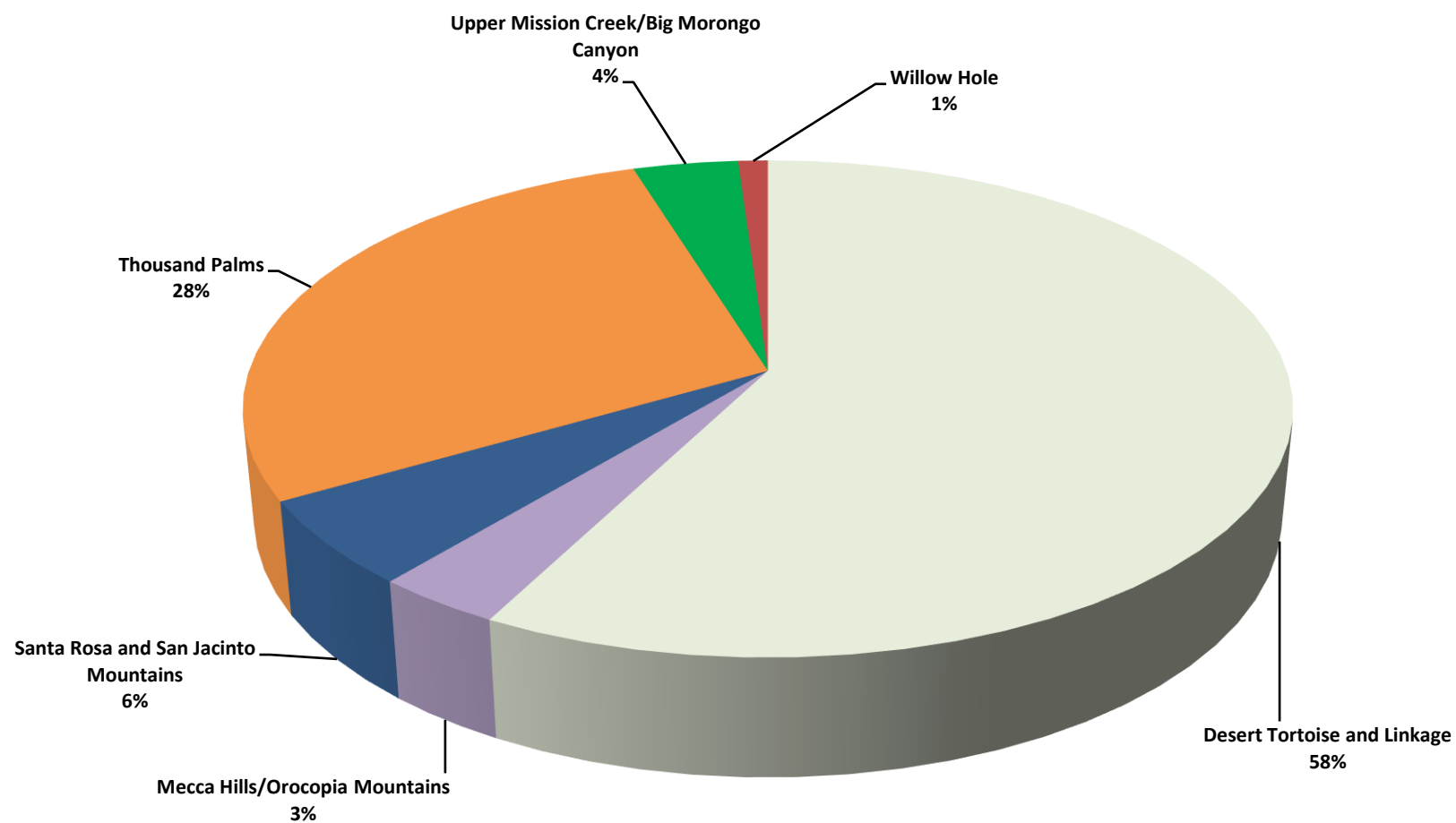


Figure 3: Total acquisitions in 2020 by Conservation Area.

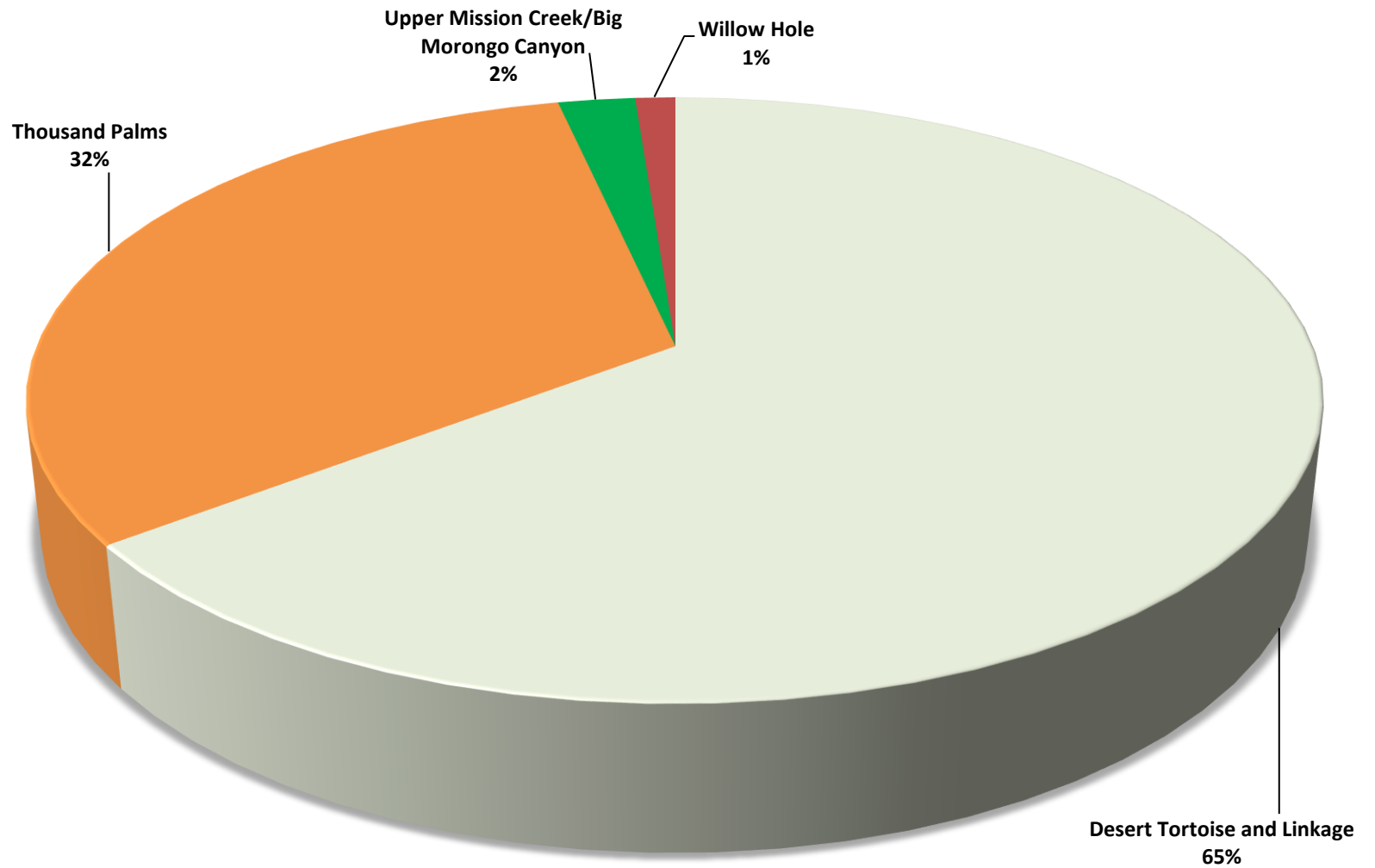


Figure 4: CVCC acquisitions in 2020 by Conservation Area.

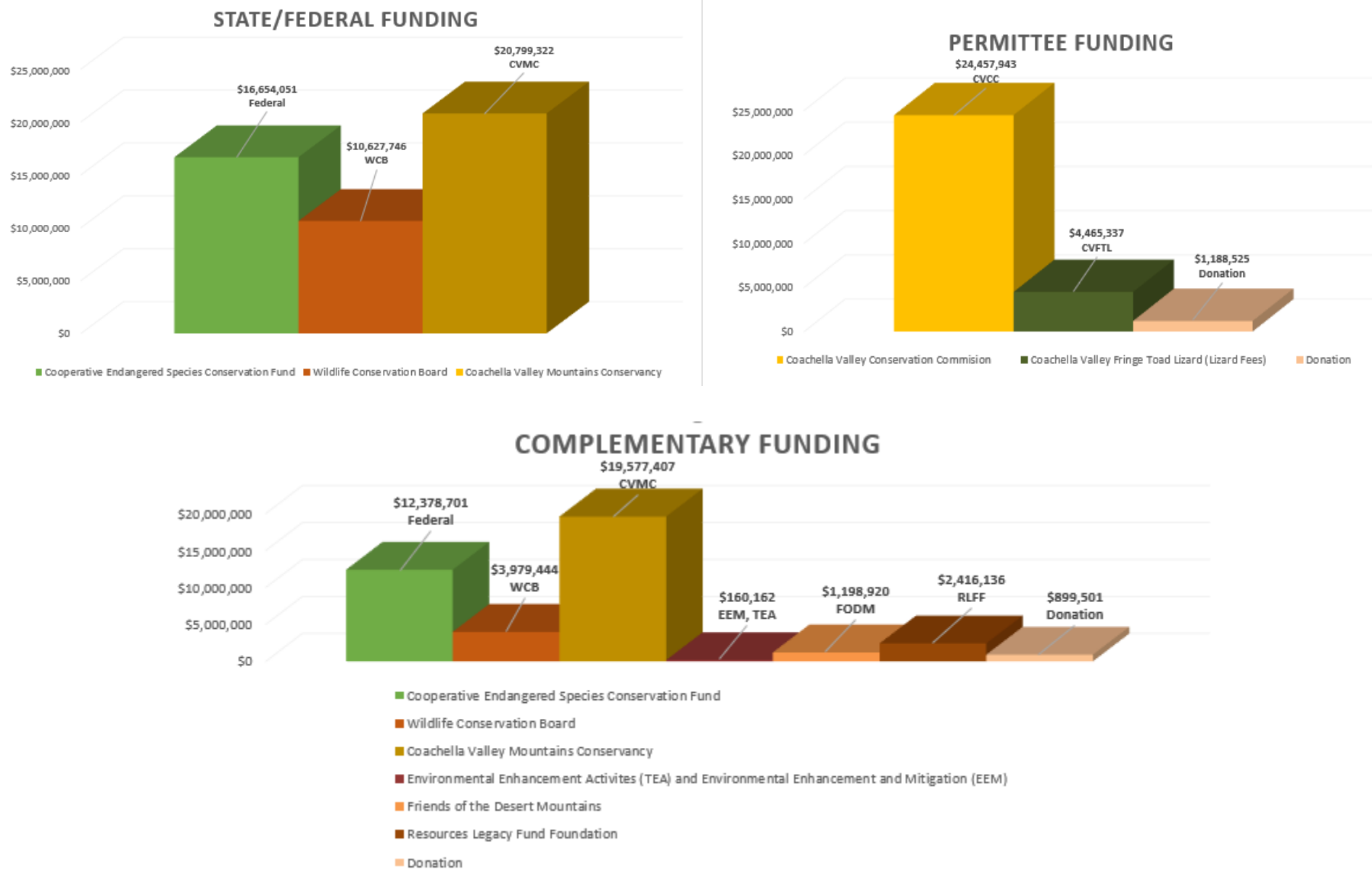


Figure 5: Acquisition funding by source.

III. Status of Conservation Areas

To ensure the persistence of the Covered Species and natural communities, the CVMSHCP includes specific acreage requirements for both the amount of authorized disturbance that can occur and the acres that must be conserved within each Conservation Area. These acreage requirements provide one measure of progress toward meeting the conservation objectives for each Covered Species, natural community, and essential ecological process in the plan. The planning process for the CVMSHCP began on November 11, 1996, and is therefore the baseline date for the authorized disturbance and conservation acreages listed throughout the CVMSHCP document.

This report updates the authorized disturbance and conservation acreages for each of the Conservation Areas through December 31, 2020 (Table 4). Authorized disturbance results from permitted development projects in the Conservation Areas while conservation occurs when land is acquired or otherwise legally protected in perpetuity by a Permittee. In 2020, there were zero acres of authorized disturbance reported, and 2,325 acres of conservation recorded. Appendix III provides a detailed accounting for all the conservation objectives per Conservation Area up through December 31, 2020.

Status of Covered Species

An overview of the status of each of the Covered Species for each Conservation Area can be found in Appendix III.

Covered Activities Outside Conservation Areas

The CVMSHCP allows for development and other Covered Activities outside the Conservation Areas which do not have to meet specific conservation objectives. A table that includes an accounting of the number of acres of Core Habitat and Other Conserved Habitat for the Covered Species and conserved natural communities that have been developed or impacted by Covered Activities outside the Conservation Areas can be found in Appendix IV. This information is listed for each of the Permittees with lands impacted by covered activities outside the Conservation Areas.

Development inside Conservation Areas has been carefully tracked and subject to review under the 1996 Memorandum of Understanding that began the planning process for the CVMSHCP. For development outside Conservation Areas, the acre figures in the table are estimates derived from the Developed area of the California Department of Conservation, Division of Land Resource Protection, Farmland Mapping and Monitoring Program GIS coverages from 1996 and 2018.

See <http://www.conservation.ca.gov/dlrp/fmmp> for more detail on the Farmland Mapping and Monitoring Program.

Table 4: Conservation and authorized disturbance within conservation areas

Conservation Area	Conservation Goal	Conserved in 2020	Conserved Since 1996	Allowed Authorized Disturbance	Authorized Disturbance in 2020	Total Authorized Disturbance since 1996
Cabazon	2,340	0	0	260	0	0
CV Stormwater Channel and Delta	3,870	0	871	430	0	5
Desert Tortoise and Linkage	46,350	1,341	6,718	5,150	0	2
Dos Palmas	12,870	0	4,283	1,430	0	0
East Indio Hills	2,790	0	35	310	0	0
Edom Hill	3,060	0	2,077	340	0	2
Highway 111/I-10	350	0	54	40	0	0
Indio Hills Palms	2,290	0	1,039	250	0	0
Indio Hills / Joshua Tree National Park Linkage	10,530	0	9,000	1,170	0	6
Joshua Tree National Park	35,600	0	13,326	1,600	0	0
Long Canyon	0	0	0	0	0	0
Mecca Hills/Orocopia Mountains	23,670	80	7,219	2,630	0	0
Santa Rosa and San Jacinto Mountains	55,890	130	32,980	5,110	0	10
Snow Creek/Windy Point	2,340	0	935	260	0	0
Stubbe and Cottonwood Canyons	2,430	0	1,057	270	0	1
Thousand Palms	8,040	662	5,181	920	0	55
Upper Mission Creek/Big Morongo Canyon	10,810	87	7,474	990	0	21
West Deception Canyon	1,063	0	1,833	100	0	0
Whitewater Canyon	1,440	0	956	160	0	1
Whitewater Floodplain	4,140	0	908	460	0	33
Willow Hole	4,920	25	2,423	540	0	6
Total	234,793	2,325	98,369	22,420	0	142

IV. Biological Monitoring Program

The CVMSHCP outlines a scientifically-based monitoring program for species, natural communities and landscapes listed under the Plan. To ensure long-term conservation goals are attained, monitoring activities are based on a three-phased approach and consist of: 1) assessing baseline conditions and identifying threats and stressors; 2) performing focused monitoring including threats and stressors, once they are determined; and 3) conducting adaptive management actions whereby the scientific method is employed to develop and implement best management practices.

The Reserve Management Unit Committee and Biological Working Group (RMUC/BWG) meet regularly to discuss updates on biological issues and adaptive management strategies. One of the tasks of these meetings is to assess current monitoring protocols to align them with research goals and management needs outlined within the CVMSHCP, as well as vetting completed monitoring activities. During the spring, the RMUC/BWG assess the monitoring priorities to be brought forth to the Reserve Management Oversight Committee as the recommended annual work plan, and each year they recommend a suite of species for monitoring that should be added in years with or following above average rainfall. The CVCC Conservation Program Manager facilitates these meetings of the Reserve Management Unit Committees and the Biological Working Group to better manage biological monitoring contracts, pursue funding opportunities for further research, and organize logistics for monitoring and land management efforts throughout the year.

To support these goals, CVCC staff actively pursue grant funding for monitoring programs. CVCC received notice of an award of funding for a project from the Natural Community Conservation Planning Local Assistance Grant (LAG) program, in the amount of \$90,672 for “Determining the distribution, status and linkages of Agassiz’s desert tortoise populations in the uplands surrounding the Coachella Valley.” The primary focus of this project is to complete an assessment of the distribution, demographics, reproductive output, and genetic linkages of desert tortoises in the foothills surrounding the Coachella Valley, especially in critical linkage areas. This will be accomplished by surveying as many parcels of conservation land as possible for the presence and abundance of desert tortoises focusing on the uplands surrounding the Coachella Valley including the San Jacinto, San Bernardino, Little San Bernardino, and Santa Rosa mountains. By identifying the presence of tortoises within these linkage areas, CVCC can better target management actions to help support recovery of the species in the Coachella Valley. This project will continue through 2021 and 2022.

In June 2020, a contract with UC Riverside (UCR) Center for Conservation Biology was approved for monitoring of aeolian sand species, burrowing owl and Palm Springs pocket mouse, Orcoplia sage, Mecca aster, triple-ribbed milkvetch, little San Bernardino mountains linanthus, and testing of a honey mesquite monitoring protocol. In coordination with the RMUC and Biological Working Group, UCR provides regular guidance and input on the development of the monitoring program tasks and performs the majority of monitoring efforts with their team of ecologists who have specialties in various aspects of the Coachella Valley desert ecology. The monitoring reports can be found in Appendices V through X respectively. The San Diego Natural History Museum (SDNHM) implemented limited monitoring among the CV Stormwater Channel and Delta Conservation Area for riparian birds. Their monitoring reports can be found along with the cowbird management report in Appendix XI. CVCC also previously contracted with the United States Geological Survey to monitor tortoise populations and demography within a focal plot south of Interstate 10 in the Desert Tortoise and Linkage Conservation Area (2019 Annual Report), using

radiotelemetry to locate the tortoises, and provide population estimates as they did previously for the population north of Interstate 10 in Cottonwood Wash (2017 Annual Report). Tortoises captured in the Desert Tortoise and Linkage Conservation Area, the Santa Rosa and San Jacinto Mountains Conservation Area and the Whitewater Canyon Conservation Area had blood taken and genetic analysis completed in 2020. Published analyses of the genetic sampling for all conservation areas is available in Appendix XII.

Peninsular bighorn sheep monitoring continued with tracking GPS telemetry collars that were fitted to sheep in the Santa Rosa and San Jacinto Mountains Conservation Area in October 2014 and November 2015. Additional GPS collars were placed on bighorn sheep in November 2017, 2018, and November 2020 funded in part by CVCC. During these bighorn captures, blood and serum samples were collected from each bighorn sheep to provide data on health and genetic status. In 2018 CDFW placed 12 additional Lotek GPS collars on bighorn in the San Jacinto and Santa Rosa Mountains Conservation Area and in November 2020, CDFW placed 15 additional collars in the Santa Rosa Mountains that were purchased by CVCC. USGS biologists have also been collecting data in 2019 and 2020 for a pilot study to examine recreational use along trails within Peninsular bighorn sheep (PBS) habitat in the Santa Rosa and San Jacinto Mountains Conservation Area, including near PBS lambing areas and watering holes. Levels of trail use were collected by using remotely deployed Trafx infrared counters operating continuously at the trailhead and interior along the trail network and scored on their heaviness of use and type of use, as recorded by infrared counters (raw counts). Recreational use was documented through long-term deployments of these trail counters that have been placed on 26 trails from 2015 to the present. Of these, 16 have collected trail use data 24 hours a day for at least a full year. From these 16 trails, biologists selected nine of these trails to represent low, medium, and high-use areas near trailheads and approximately one kilometer from trailheads. After the initial analysis, BLM deployed 15 more infrared counters for further analysis of their use, bringing the total to 35 active counters throughout the study area. Observers in the form of citizen scientists were also deployed to test the protocol during lambing season in winter and spring of 2019/2020. Once the protocol was developed, several human observers (citizen scientists) were deployed to document behavior (running, jumping, shouting) and record the numbers of dogs, bikes, up-and-back hikers, and other types of recreational users that the counters cannot detect. Due to significant setbacks in deployment of volunteers and continuation of the study through the Stay-at-Home orders, the final report for this study will be included in the 2021 Annual Report.

2020 Biological Monitoring Activities



Photos: 1– Burrowing owllets in natural nest burrow; 2 – Mecca aster transect; 3 – Dr. Lovich of USGS documents a male tortoise

V. Land Management Program

Management of lands acquired by CVCC and other local Permittees is coordinated with management of the existing conservation lands owned by state, federal and non-profit agencies. The Reserve Management Oversight Committee (RMOC) is the inter-agency group that provides a forum for coordination of management and monitoring lands within the Reserve System and makes recommendations to the CVCC. The Reserve Management Oversight Committee is supported by the Reserve Management Unit Committees.

Due to the stay at home orders to combat the COVID-19 pandemic, the Reserve Management Oversight Committee held one in person meeting January 22, 2020 and two Zoom meetings May 8 and October 28, 2020. Each RMOC meeting included a report regarding the Monitoring Program and the Land Management Program. The RMOC reviewed the Reserve Management and Monitoring work plans, biological monitoring and management priority activities, and tentative budget remotely in May, due to the delay in scheduling from the COVID-19 pandemic. The recommendations from the RMOC were incorporated into the CVCC budget for FY 2020/21 and presented to the CVCC at their June 2020 meeting by Zoom. CVCC staff continues to coordinate with the RMOC and RMUCs to ensure that monitoring and research activities inform and support management of the Reserve Management Units.

Reserve Management Unit Committees

The six Reserve Management Units (RMUs) facilitate coordinated management by local, state and federal agencies to achieve the Conservation Objectives within the MSHCP Reserve System. Because many of the same staff members are involved in both the RMUC and BWG, meetings were combined to reduce demands on staff time and provide for better coordination between management and monitoring teams. The RMUC/BWG met by Zoom on March 10, September 8, and December 8, 2020. The group discussed prioritizing invasive species and off-road vehicle control management efforts, increasing volunteer opportunities, priorities for monitoring and research, coordination on grant opportunities, and monitoring results. With a focus on invasive species management, members also met with the Low Desert Weed Management Area for CalFlora Weed Manager training on January 23, 2020 and for a regional meeting December 3, 2020.

Trails Management Subcommittee

The Trails Management Subcommittee (TMS) meetings were held in person on January 15, 2020 and by Zoom on May 20, September 16, and November 18, 2020. Working groups in 2020 included a focus on Dog Enforcement and Ordinances, Trail Maintenance, and Trails Research. The TMS working groups report on progress for their tasks and discuss significant issues, management, and funding opportunities at the quarterly TMS meetings. When possible due to the Stay-at-Home order, Friends of the Desert Mountains and their volunteer crew continued to worked closely with BLM and the cities to fix trail hazards and install clear directional and safety signage. Friends' volunteers are taking the lead on trail restoration throughout the valley. CVCC staff also worked with the Oswit Canyon Land Trust, Friends of the Desert Mountains, and other volunteers from the TMS to install "No Dogs On Trails" signs in the City of Palm Springs, and support interpretive rangers at trailheads to increase awareness of why the No Dog ordinance was passed for the protection of bighorn sheep.

Land Improvement: Acquisition Cleanups

In 2020 the CVCC Acquisitions Manager performed pre-acquisition site inspections and job walks on 30 parcels/projects in multiple Conservation Areas. During these inspections the Land Acquisitions Manager identified illegal dumping, hazardous conditions, OHV & equestrian activity, and the existence of listed species, as well as determined property fencing requirements. As per CVCC's standard Purchase & Sale Agreements, willing sellers are required to clean up illegal dumping and blight prior to closing. Contractors are met in the field by the Acquisitions Manager prior to a required cleanup to review the agency's standards and specifications for the particular site in question. After cleanup, the job site is re-inspected to certify that cleanups meet the requirements, and if they are found lacking, the seller is notified if additional work will be necessary. After closing, CVCC monitors the sites at least annually for ongoing management/fencing requirements. This year, CVCC was directly responsible for removing an estimated 2.88 tons of refuse, including 310 tires, from the Coachella Valley, covering more than 678.66 acres and generating over \$38,350 in contractor revenue from sellers' property sales.

Property Management & Monitoring

Monitoring the status of CVCC conservation lands is an essential and ongoing activity. Site visits and patrols were minimized during the Stay-at-Home orders of 2020 and resumed to monthly visits in the fall and winter to various CVCC properties. Illegal dumping, OHV use and shooting continue to be a problem on some of the Reserve lands, and these issues were widely exacerbated with the lack of law enforcement inability to respond to non-emergency issues during the height of the pandemic. In January 2020, CVCC's fencing contractor installed 10,602 linear feet of post and cable fencing, signs and gates to protect mesquite dunes in the West Deception Canyon Conservation Area. Working in partnership to secure adjacent conservation lands, the Coachella Valley Mountains Conservancy paid to clean up and fence adjacent CVCC and Friends of the Desert Mountains properties and reimbursed CVCC \$86,324.70 for the fencing project. CVCC also maintained fencing and signage installed previously within the Willow Hole, Upper Mission Creek and Big Morongo Canyon, Sky Valley, and Stubbe and Cottonwood Canyon Conservation Areas. By the end of 2020, CVCC had 16.4 miles of the fencing around the perimeter of the Conservation Lands to monitor and maintain. The gap in law enforcement and monitoring of the fencing and gates required an increase in patrols once the stay-at-home orders were lifted due to the increase in dumping or OHV activity in conservation areas. CVCC also began working on contracting local law enforcement in Desert Hot Springs to monitor and cite illegal activity in conservation lands in and just adjacent to the City boundaries.

In addition to fencing and signage, CVCC staff worked with the Urban Conservation Corps and Coachella Valley Mountains Conservancy to control invasive vegetation on properties in the Willow Hole and Stormwater Channel and Delta Conservation Area. CVCC finished removing tamarisk from the Willow Hole area in 2020. CVCC also received funding for a project in late 2019 from the Coachella Valley Mountains Conservancy through their Proposition 1 Water Bond (Prop 1) program, in the amount of \$295,974 for "Wetlands Restoration, Tamarisk Control and Rail Habitat Enhancement Project." This project has initiated restoration of the North Shore Ranch property in the Coachella Valley Stormwater Channel and Delta Conservation Area. This project will focus on removal of non-native tamarisk and development of a wetlands restoration and enhancement plan for rails and mesquite habitat. Tamarisk removal began in January 2020 on the properties, with assistance from the Southern California Mountains Foundation Urban Conservation Corps. CVCC also contracted with GPA Consulting to develop a draft restoration plan for Pond 5 on the property in November 2020.

CVCC contracted with the San Diego Natural History Museum again in 2020 to continue to control invasive cowbirds in the Coachella Valley Stormwater Channel and Delta, and Dos Palmas Conservation Areas. The 2020 Cowbird Management Report can be found in Appendix XI.

2020 Land Management Activities



Photos: 1 – Cowbird trap in Coachella Valley Stormwater Channel and Delta Conservation Area; 2 – Cowbird collected from trap; 3 – Urban Conservation Corps remove invasive tamarisk from North Shore Ranch in the Stormwater Channel and Delta Conservation Area; 4- Urban Conservation Corps remove large tamarisk from mesquite dune area; 5- New fencing and signage in West Deception Canyon; 6- Invasive stinknet found next to Worsley Rd. in Desert Hot Springs.

VI. Unauthorized Activities and Enforcement

Off-highway vehicles, dumping and vandalism of fencing continue to be issues. In 2020, areas where these problems were reported included Stubbe/Cottonwood Canyon, Willow Hole, Upper Mission Creek/Big Morongo Canyon, and Thousand Palms Conservation Areas. Further discussion of management of these issues is included in section IV. Currently CVCC forwards reports of OHVs and dumping to the appropriate law enforcement agency. CVCC is working to develop an agreement with Sheriffs Department under which CVCC would contribute funds to hire additional law enforcement deputies to focus on the illegal activity in Conservation Areas.

VII. Significant Issues in Plan Implementation

In 2020, progress continued on the La Quinta Peninsular Bighorn Sheep Barrier Project. The primary objective of the La Quinta Peninsular Bighorn Sheep Barrier Project is to protect PBS by preventing them from accessing and coming to harm from using urban lands, including golf courses and landscaping, artificial water bodies, and roadways. The CVCC certified the Final Environmental Impact Report for the La Quinta Peninsular Bighorn Sheep Barrier Project, adopting findings and a Mitigation Monitoring and Reporting program, and approving Alternative A2 of the La Quinta Peninsular Bighorn Sheep Barrier Project in April 2019. A competitive sealed-bid process was completed, with four bids being received by the deadline of January 13, 2020. A contract with American Fence Co., Inc. to install the fence was approved by the CVCC at the February 13, 2020 meeting for a not to exceed amount of \$2,091,656.58. Since CVCC does not control the land needed for a fence, an ongoing effort has been to work with property owners on access agreements for construction and permanent installation. During 2020, staff continued work to complete agreements for use of property owned or managed by PGA West, a license agreement with the Bureau of Reclamation (Reclamation), encroachment permits from Coachella Valley Water District, and use permit from Riverside County Regional Parks and Open Space District (for the Lake Cahuilla Veterans Memorial Park). Good progress was made in getting the necessary documents in place with CVWD, Riverside County, and Reclamation; the next step for these permits is completion of construction plans. CVCC staff has been working with PGA West management to obtain a license agreement with PGA West owners to allow for access to allow fence construction. In fall 2020, PGA West notified CVCC of their concerns about the potential for the fence route to impact views. PGA West indicated that access to their property will not be granted until they are satisfied that the visual impacts of the fence have been reduced as much as possible. CVCC staff conducted a site visit with PGA West management and homeowners' association representatives to review options and identify ways to reduce visual impacts of the fence. Additional work on preparing for installation of the fence, including completion of construction plans, is pending resolution and an access agreement with PGA West in 2021.

Progress was made in 2020 on another significant challenge on CVCC lands, control of illegal activities including vandalism, illegal OHV use, and dumping. Since CVCC does not have law enforcement capacity, the intent has been to develop an agreement with one or more of the local agencies that have trained law enforcement personnel. In April 2020, a services contract was signed with a consultant who is assisting CVCC staff with negotiations with local law enforcement agencies to develop such agreements. The concept was to develop an MOU which would provide for coordination of certain management activities to ensure increased protection of lands within the CVMSHCP Reserve System. Because CVCC has many properties in CVMSHCP Conservation Areas that are within or adjacent to the Desert Hot Springs city limits, outreach to the City was initiated in fall 2020 to identify opportunities for collaboration and develop a Memorandum of Understanding (MOU). A final MOU would be developed for consideration by

CVCC and Desert Hot Springs in early 2021. Efforts to secure MOUs with other law enforcement agencies including Riverside County Sheriff's Department would follow. This approach will result in cooperative and coordinated management that will create efficiencies and enhance each entity's management capabilities and effectiveness.

VIII. Compliance Activities of Permittees

All Permittees are in compliance with requirements of the CVMSHCP. CVCC completed six Joint Project Reviews for Permittees in 2020.

All the cities are complying with the fee language in their ordinances. All jurisdictions report their Local Development Mitigation Fee (LDMF) activity and remit the revenue to CVCC monthly. CVCC reviews all LDMF reports and receipts. The LDMF generated a total of \$2,691,023 in fiscal year 2019/2020. This represents a 40 percent reduction from \$4,454,268 that was generated in fiscal year 2018/19. However, it should be noted that most of the previous fiscal year's revenue was generated by CVCC's highest LDMF payment ever for a single project: \$2,664,978, for a 2,140-acre photovoltaic power project. By way of comparison, the reported revenues for fiscal year 2019/2020 are 19 percent higher than the revenues generated in fiscal year 2017/2018.

IX. Expenditures for CVMSHCP - 2020/21 Budget

Full budget available at:

https://www.cvag.org/library/pdf_files/admin/CVCC%20Financials%20Reports%20FY_2020_2021/CVCC_budget2021.pdf

BUDGET BY PROGRAMS - FY 2020/2021

	MANAGEMENT AND MONITORING	GENERAL ADMINISTRATION	LAND ACQUISITION	ENDOWMENT	LIZARD ENDOWMENT	TRAVERTINE MANAGEMENT	MANAGEMENT CONTINGENCY	IN-LIEU FEE	TOTAL
BEGINNING FUND BALANCE	\$ 527,590	\$ 697,323	\$ 7,931,378	\$ 8,563,068	\$ 329,705	\$ 538,459	\$ 4,869,759	\$ 1,001,662	\$ 24,458,944
REVENUES:									
Development Mitigation Fees	\$ 253,601	\$ -	\$ 1,238,169	\$ -	\$ -	\$ -	\$ -	\$ -	\$ 1,491,770
Agencies Mitigation Fees	-	-	1,500,000	1,000,000	-	-	-	-	2,500,000
Tipping Fees	-	440,000	-	-	-	-	-	-	440,000
Contributions	-	-	-	-	-	-	-	-	-
Grants	240,407	-	-	-	-	-	-	-	240,407
Other Revenue	7,500	-	-	-	-	-	-	-	7,500
Investment Income	8,450	3,850	82,615	79,125	3,300	5,390	50,275	100	233,105
Total Revenues	\$ 509,958	\$ 443,850	\$ 2,820,784	\$ 1,079,125	\$ 3,300	\$ 5,390	\$ 50,275	\$ 100	\$ 4,912,782
EXPENDITURES:									
Administrative Fees	\$ 2,536	\$ -	\$ 12,382	\$ -	\$ -	\$ -	\$ -	\$ -	\$ 14,918
Accounting / Bank Service Charges	-	4,305	-	-	-	-	-	1,785	6,090
Comprehensive Insurance	-	13,442	-	-	-	-	-	-	13,442
Per Diem Payments	-	10,200	-	-	-	-	-	-	10,200
Per Diem Taxes	-	882	-	-	-	-	-	-	882
Office Supplies	3,000	2,000	-	-	-	-	-	-	5,000
Printing	-	2,000	-	-	-	-	-	-	2,000
Land Improvements	2,103,816	-	500,000	-	-	-	-	-	2,603,816
Legal Services	2,000	30,000	-	-	-	-	-	-	32,000
Professional Services	4,000	10,200	170,000	-	-	-	-	-	184,200
Consultants (Regular funds)	818,854	262,506	351,897	-	-	-	-	54,164	1,487,421
Consultants (Grant funds)	240,407	-	-	-	-	-	-	-	240,407
Miscellaneous	700	1,000	500	-	-	-	-	200	2,400
Land Acquisitions	-	-	8,500,000	-	-	-	-	-	8,500,000
Furniture and Equipment	-	-	-	-	-	-	-	-	-
Utilities	15,000	-	-	-	-	-	-	-	15,000
Sub-Total Expenditures	\$ 3,190,313	\$ 336,535	\$ 9,534,779	\$ -	\$ -	\$ -	\$ -	\$ 56,149	\$ 13,117,776
OTHER									
Operating Transfers Out	\$ -	\$ -	\$ -	\$ 300,014	\$ -	\$ -	\$ 2,103,816	\$ -	\$ 2,403,830
Operating Transfers In	(2,403,830)	-	-	-	-	-	-	-	(2,403,830)
Sub-Total Other	\$ (2,403,830)	\$ -	\$ -	\$ 300,014	\$ -	\$ -	\$ 2,103,816	\$ -	\$ -
Total Expenditures and Other	\$ 786,483	\$ 336,535	\$ 9,534,779	\$ 300,014	\$ -	\$ -	\$ 2,103,816	\$ 56,149	\$ 13,117,776
Net Excess (Deficit)	\$ (276,525)	\$ 107,315	\$ (6,713,995)	\$ 779,111	\$ 3,300	\$ 5,390	\$ (2,053,541)	\$ (56,049)	\$ (8,204,994)
ENDING FUND BALANCE	\$ 251,065	\$ 804,638	\$ 1,217,383	\$ 9,342,179	\$ 333,005	\$ 543,849	\$ 2,816,218	\$ 945,613	\$ 16,253,950

X. Annual Audit

CVCC approved their Fiscal Year 2020/2021 budget at the June 11, 2020 meeting.

The audit of the expenditures for the period July 1, 2019 to June 30, 2020 was approved by CVCC on March 11, 2020. The financial report was designed to provide citizens, members, and resource providers with a general overview of the CVCC's finances, and to show accountability for the money it receives. Questions about this report or additional financial information can be obtained by contacting the CVCC Auditor, at 73-710 Fred Waring Drive, Suite 200, Palm Desert, CA 92260. Annual CVCC audits are available at http://www.cvag.org/cvcc_financial_reports.htm.

XI. In-Lieu Fee Program

In 2014, CVCC completed the Enabling Instrument for an In-Lieu Fee Program (ILFP) with the U.S. Army Corps of Engineers (USACE). Much like the CVM SHCP, an ILFP replaces piecemeal mitigations with a coordinated approach that complements other conservation efforts by allowing developers to offset impacts to Waters of the United States through the purchase of fee credits that will support a targeted restoration project. CVCC has identified a site within the Coachella Valley Stormwater Channel and Delta Conservation Area as a potential restoration project, and is currently awaiting approval from USACE. In the meantime, the Corps has authorized CVCC to sell up to 50 advance credits to generate seed funding for the restoration project. Developers may be directed to purchase credits through sections 401 and 404 of the federal Clean Water Act, or through section 1602 of the California Fish and Game Code. Credits are priced to accommodate both the restoration and long term management of one acre of Waters of the United States.

Since May 2016, CVCC has sold 7.3 advance credits (Table 5). While restoration is generally expected to begin within three growing seasons of the first credit sale, CVCC has entered into an Instrument Modification with USACE to extend the interim period while the final restoration site is determined.

Table 5: In-Lieu Fee Program advance credit purchases.

Applicant	Mitigation Type	Credits Purchased	Date of Purchase
Palm Springs	Restoration	0.35	31-05-16
Caltrans	Enhancement	0.18	02-12-16
Riverside County	Enhancement	0.03	05-09-17
Southern California Edison	Restoration	1.26	28-03-18
Riverside County	Enhancement	0.199	25-05-18
Caltrans	Enhancement	0.498	16-09-18
Indio	Enhancement	1.00	21-10-19
CVAG	Restoration	2.8	07-11-19
Terra-Gen	Restoration	0.36	19-05-20
Terra-Gen	Restoration	0.62	15-06-20
Total		7.3	

Appendix I

Rules for Land Acquisition and Management Credit

Acquisition Credit

In general, the source of funds for acquisition gets the credit of acres with the following modifications:

- 1) Per Plan Section 4.2.1 (p. 4-10), purchases with state or federal funding will be considered Complementary in the following Conservation Areas: Joshua Tree National Park, the Santa Rosa and San Jacinto Mountains, the Mecca Hills and Orocopia Mountains, and Snow Creek/Windy Point. Purchases within these areas with CVCC funds will be considered Permittee.
 - a. If land purchased with non-federal/state funding in these areas is transferred to CVCC ownership, it will be considered a donation and CVCC will receive Permittee credit if they take title. Examples include:
 - i. Purchases by Friends of Desert Mountains (FODM) – only if funds are from private foundations (e.g. Resources Legacy Fund);
 - ii. Donations from landowners.
- 2) Acquisitions in Fluvial Sand Transport Only Areas will be credited to the funding entity (Permittee, Complementary, and Federal/State).
 - a. If federal/state funds will be counted as federal/state acquisition
 - b. If land purchased with non-federal/state funding in these areas is transferred to CVCC, it will be considered a donation and CVCC will receive Permittee credit.
- 3) For 2015 Annual Report parcels adjacent to Conservation Areas will not be counted but will be included in the overall database and flagged for consideration after the issue of a legal instrument for conservation is resolved.
- 4) If a grant requires a matching amount, that portion of the grant will be credited to the source of the match. This includes cash contributions and in-kind contributions from bargain sales (not addressed in the plan). However, as “mitigation” cannot be used as a match for Section 6 grants, Permittees cannot receive acre credit for Section 6 matches.
- 5) Mitigation for projects outside Plan Area (Wildlands, Inc. is the only current example ~ 7,000 acres) or mitigation for project not Covered as part of the Plan (Southern California Edison purchase of the mitigation value of CVCC in 2014) are included in the database but are zero for all credit and noted “conserved but it does not count for the Annual Report or Plan acreage numbers.”
- 6) No Acres within any Tribal Land are counted for the CVMSHCP under any circumstances as Tribal Land is “Not A Part” of the CVMSHCP Plan Area.

Appendix II

Table of Acquisitions for Conservation in 2020

CVMSHCP Annual Report 2020 - Parcels Acquired for Conservation

Conservation Area	Acq_Agency	APN	Total
Desert Tortoise and Linkage Conservation Area	Coachella Valley Conservation Commission	707400015	320.80
		707430002	232.25
		707430003	47.09
		707430004	36.59
		709370004	115.36
		709480007	20.04
		709500011	10.02
		709510017	19.97
		709520010	20.08
		715140030	22.47
		715140036	3.04
		743320008	313.10
		745310002	99.69
		745330003	40.12
		745330004	40.13
	Coachella Valley Conservation Commission Total	1,340.75	
Desert Tortoise and Linkage Conservation Area Total			1,340.75
Mecca Hills/Orocopia Mountains Conservation Area	Friends of the Desert Mountains	717160002	29.93
		717160005	19.97
		717160009	5.00
		717160010	5.00
		717160013	20.00
	Friends of the Desert Mountains Total	79.90	
Mecca Hills/Orocopia Mountains Conservation Area Total			79.90
Santa Rosa and San Jacinto Mountains Conservation Area	Coachella Valley Conservation Commission	773030006	48.55
	Coachella Valley Conservation Commission Total		48.55
	Friends of the Desert Mountains	636092013	1.75
		636092014	1.59
		753330014	39.42
		753330015	39.07
	Friends of the Desert Mountains Total	81.83	
Santa Rosa and San Jacinto Mountains Conservation Area Total			130.39
Thousand Palms Conservation Area	Coachella Valley Conservation Commission	648060001	646.28
		648110006	5.08
		648160013	2.37
		752010005	8.40
	Coachella Valley Conservation Commission Total	662.13	
Thousand Palms Conservation Area Total			662.13
Upper Mission Creek/Big Morongo Canyon Conservation Area	Coachella Valley Conservation Commission	664060032	3.10
		664120006	5.41
		664130009	18.00
		667020009	15.59
		667020014	5.15
	Coachella Valley Conservation Commission Total		47.26
	Friends of the Desert Mountains	638060001	39.97
	Friends of the Desert Mountains Total		39.97
Upper Mission Creek/Big Morongo Canyon Conservation Area Total			87.23
Willow Hole Conservation Area	Coachella Valley Conservation Commission	659220005	4.60
		659230042	5.05
		659240003	4.99
		659250012	4.95
		660340008	5.01
Coachella Valley Conservation Commission Total	24.59		
Willow Hole Conservation Area Total			24.59
Grand Total			2,324.98

Appendix III

Status of Conservation Objectives by Conservation Area

CVMSHCP Annual Report 2020 - Conservation Objectives by Conservation Area

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Cabazon Conservation Area - Riverside County								
Peninsular Bighorn Sheep - Essential Habitat	264	181	83	0	0	0%	0	18
Mesquite hummocks	13	1	12	0	0	0%	0	0
Southern sycamore-alder riparian woodland	9	1	9	0	0	0%	0	0
Sand Source	7,683	181	1,629	0	0	0%	0	18
Sand Transport	4,538	0	0	0	0	0%	0	0
Fornat Wash Corridor	641	10	631	0	0	0%	0	1
Coachella Valley Stormwater Channel and Delta Conservation Area - Riverside County								
Desert Pupfish - Core Habitat	25	0	25	0	0	0%	0	0
Crissal Thrasher - Core Habitat	896	87	781	371	0	48%	5	41
California Black Rail - Other Conserved Habitat	62	6	52	0	0	0%	0	1
Yuma Clapper Rail - Other Conserved Habitat	62	6	52	0	0	0%	0	1
Le Conte's Thrasher - Other Conserved Habitat	784	78	706	371	0	53%	5	40
Mesquite hummocks	74	7	67	20	0	30%	0	3
Coastal and valley freshwater marsh	61	6	51	0	0	0%	0	1
Desert sink scrub	1,349	114	1,026	44	0	4%	0	16
Desert saltbush scrub	792	79	713	351	0	49%	5	38

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Desert Tortoise and Linkage Conservation Area - Coachella								
Desert Tortoise - Core Habitat	300	30	270	0	0	0%	0	3
Le Conte's Thrasher - Other Conserved Habitat	300	30	270	0	0	0%	0	3
Desert dry wash woodland	121	12	109	0	0	0%	0	1
Desert Tortoise and Linkage Conservation Area - Riverside County								
Desert Tortoise - Core Habitat	88,878	4,998	44,977	6,415	1,341	14%	16	1,125
Orocopia Sage - Core Habitat	779	44	398	0	0	0%	0	4
Mecca Aster - Core Habitat	4,731	206	1,855	333	0	18%	0	54
Le Conte's Thrasher - Other Conserved Habitat	49,114	2,813	25,319	2,386	782	9%	16	504
Desert dry wash woodland	13,443	752	6,771	874	233	13%	8	155
Desert Tortoise and Linkage Corridor	26,122	1,572	14,143	2,010	662	14%	0	358

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Dos Palmas Conservation Area - Riverside County								
Crissal Thrasher - Core Habitat	536	38	343	235	0	69%	0	27
Desert Pupfish - Refugia Locations	0	0	0	0	0	0%	0	0
California Black Rail - Other Conserved Habitat	597	37	334	281	0	84%	0	32
Le Conte's Thrasher - Other Conserved Habitat	14,882	743	6,689	2,460	0	37%	0	320
Yuma Clapper Rail - Other Conserved Habitat	682	42	374	301	0	80%	0	35
Predicted Flat-tailed Horned Lizard - Other Conserved Habitat	5,537	403	3,631	681	0	19%	0	108
Desert fan palm oasis woodland	125	6	50	29	0	59%	0	4
Arrowweed scrub	277	13	121	0	0	0%	0	1
Mesquite bosque	482	36	320	225	0	70%	0	26
Desert sink scrub	7,195	487	4,381	1,179	0	27%	0	167
Desert dry wash woodland	1,856	83	746	245	0	33%	0	33
Cismontane alkali marsh	321	23	205	200	0	98%	0	22
Mesquite hummocks	55	3	23	12	0	51%	0	2
East Indio Hills Conservation Area - Coachella								
Le Conte's Thrasher - Other Conserved Habitat	62	6	56	0	0	0%	0	1
Palm Springs Pocket Mouse - Other Conserved Habitat	8	1	7	0	0	0%	0	0
Coachella Valley Round-tailed Ground Squirrel - Other Conserved Habitat	6	1	5	0	0	0%	0	0
Predicted Flat-tailed Horned Lizard - Other Conserved Habitat	6	1	5	0	0	0%	0	0

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
East Indio Hills Conservation Area - Indio								
Le Conte's Thrasher - Other Conserved Habitat	120	12	105	0	0	0%	0	1
Palm Springs Pocket Mouse - Other Conserved Habitat	117	11	103	0	0	0%	0	1
Coachella Valley Round-tailed Ground Squirrel - Other Conserved Habitat	117	11	103	0	0	0%	0	1
Predicted Flat-tailed Horned Lizard - Other Conserved Habitat	114	11	100	0	0	0%	0	1
Mesquite hummocks	2	0	2	0	0	0%	0	0
Stabilized shielded sand fields	114	11	100	0	0	0%	0	1
East Indio Hills Conservation Area - Riverside County								
Le Conte's Thrasher - Other Conserved Habitat	1,960	139	1,253	35	0	3%	0	17
Mecca Aster - Core Habitat	1,594	116	1,045	0	0	0%	0	12
Coachella Valley Round-tailed Ground Squirrel - Other Conserved Habitat	1,353	100	896	1	0	0%	0	10
Predicted Flat-tailed Horned Lizard - Other Conserved Habitat	525	46	415	0	0	0%	0	5
Palm Springs Pocket Mouse - Other Conserved Habitat	1,526	105	944	33	0	3%	0	14
Active desert dunes	5	1	4	0	0	0%	0	0
Desert saltbush scrub	8	1	7	0	0	0%	0	0
Stabilized desert sand fields	331	33	295	0	0	0%	0	3
Mesquite hummocks	43	4	39	0	0	0%	0	0
Stabilized shielded sand fields	401	28	256	0	0	0%	0	3

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Edom Hill Conservation Area - Cathedral City								
Coachella Valley Round-tailed Ground Squirrel - Other Conserved Habitat	134	13	121	102	0	84%	0	11
Coachella Valley Milkvetch - Other Conserved Habitat	151	15	136	102	0	75%	0	12
Palm Springs Pocket Mouse - Other Conserved Habitat	114	11	103	87	0	84%	0	9
Le Conte's Thrasher - Other Conserved Habitat	344	34	310	224	0	72%	0	26
Sand Source	345	34	310	224	0	72%	0	26
Edom Hill Conservation Area - Riverside County								
Coachella Valley Giant Sand-treader Cricket - Other Conserved Habitat	103	5	40	43	0	100%	0	5
Coachella Valley Milkvetch - Other Conserved Habitat	1,637	134	1,205	1,029	0	85%	0	116
Coachella Valley Fringe-toed Lizard - Other Conserved Habitat	103	5	40	43	0	100%	0	5
Coachella Valley Round-tailed Ground Squirrel - Other Conserved Habitat	1,701	145	1,302	1,115	0	86%	0	126
Palm Springs Pocket Mouse - Other Conserved Habitat	1,228	104	935	794	0	85%	0	90
Le Conte's Thrasher - Other Conserved Habitat	2,238	194	1,745	1,334	0	76%	1	152
Active sand fields	73	4	37	41	0	100%	0	4
Stabilized desert sand fields	29	1	3	2	0	67%	0	1
Sand Source	2,665	197	1,770	1,468	0	83%	0	167
Sand Transport	628	63	565	377	0	67%	1	43

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Highway 111/I-10 Conservation Area - Riverside County								
Coachella Valley Round-tailed Ground Squirrel - Other Conserved Habitat	389	39	350	54	0	15%	0	9
Coachella Valley Jerusalem Cricket - Other Conserved Habitat	372	37	335	51	0	15%	0	9
Le Conte's Thrasher - Other Conserved Habitat	389	39	350	54	0	15%	0	9
Coachella Valley Milkvetch - Other Conserved Habitat	372	37	335	51	0	15%	0	9
Palm Springs Pocket Mouse - Other Conserved Habitat	389	39	350	54	0	15%	0	9
Indio Hills Palms Conservation Area - Riverside County								
Mecca Aster - Core Habitat	6,091	255	2,290	1,039	0	45%	0	130
Le Conte's Thrasher - Other Conserved Habitat	106	1	7	0	0	0%	0	0
Desert fan palm oasis woodland	93	5	42	7	0	17%	0	1
Desert dry wash woodland	79	4	33	36	0	100%	0	4
Mesquite hummocks	3	1	1	0	0	0%	0	0
Indio Hills/Joshua Tree National Park Linkage Conservation Area - Riverside County								
Desert Tortoise - Core Habitat	10,308	859	7,735	6,557	0	85%	0	741
Le Conte's Thrasher - Other Conserved Habitat	6,396	606	5,457	5,469	0	100%	0	607
Sand Transport	7,304	681	6,132	5,791	0	94%	5	642
Sand Source	5,823	460	4,135	3,205	0	77%	0	367
Indio Hills / Joshua Tree National Park Corridor	13,127	1,141	10,267	8,996	0	88%	5	1,009

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Joshua Tree National Park Conservation Area - Riverside County								
Gray Vireo - Other Conserved Habitat	30,653	134	1,208	1,822	0	100%	0	195
Le Conte's Thrasher - Other Conserved Habitat	4,330	25	222	104	0	47%	0	13
Desert Tortoise - Core Habitat	127,161	1,708	15,367	12,690	0	83%	0	1,440
Desert dry wash woodland	2,195	13	119	192	0	100%	0	20
Mojave mixed woody scrub	57,099	800	7,195	6,349	0	88%	0	715
Desert fan palm oasis woodland	5	0	0	0	0	0%	0	0
Mojavean pinyon & juniper woodland	30,653	134	1,208	1,822	0	100%	0	195
Mecca Hills/Orocopia Mountains Conservation Area - Riverside County								
Desert Tortoise - Core Habitat	112,575	2,624	23,617	6,873	80	29%	0	950
Le Conte's Thrasher - Other Conserved Habitat	17,467	652	5,866	1,401	0	24%	0	205
Orocopia Sage - Core Habitat	66,180	1,803	16,227	4,376	0	27%	0	618
Mecca Aster - Core Habitat	31,655	465	4,181	1,302	80	31%	0	177
Desert fan palm oasis woodland	1	0	0	0	0	0%	0	0
Desert dry wash woodland	9,317	318	2,861	1,199	9	42%	0	152
Santa Rosa and San Jacinto Mountains Conservation Area - Cathedral City								
Desert Tortoise - Other Conserved Habitat	107	11	95	0	0	0%	0	1
Le Conte's Thrasher - Other Conserved Habitat	13	1	11	0	0	0%	0	0
Peninsular Bighorn Sheep - Rec Zone 2 - Essential Habitat	112	11	97	0	0	0%	0	1
Desert dry wash woodland	20	2	18	0	0	0%	0	0

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Santa Rosa and San Jacinto Mountains Conservation Area - Indian Wells								
Desert Tortoise - Other Conserved Habitat	4,375	111	999	36	0	4%	0	15
Le Conte's Thrasher - Other Conserved Habitat	419	23	206	0	0	0%	0	2
Peninsular Bighorn Sheep - Rec Zone 3 - Essential Habitat	4,617	114	1,158	36	0	3%	0	15
Desert dry wash woodland	128	7	66	0	0	0%	0	1
Santa Rosa and San Jacinto Mountains Conservation Area - La Quinta								
Desert Tortoise - Other Conserved Habitat	5,936	157	1,409	423	48	30%	7	51
Le Conte's Thrasher - Other Conserved Habitat	683	43	387	125	3	32%	0	17
Peninsular Bighorn Sheep - Rec Zone 3 - Essential Habitat	6,185	159	2,545	439	48	17%	0	41
Desert dry wash woodland	147	8	76	15	0	20%	0	2
Santa Rosa and San Jacinto Mountains Conservation Area - Palm Desert								
Le Conte's Thrasher - Other Conserved Habitat	43	4	33	0	0	0%	0	0
Desert Tortoise - Other Conserved Habitat	581	48	436	783	0	100%	0	82
Peninsular Bighorn Sheep - Rec Zone 3 - Essential Habitat	78	7	65	0	0	0%	0	1
Peninsular Bighorn Sheep - Rec Zone 2 - Essential Habitat	492	7	65	761	0	100%	0	74
Desert dry wash woodland	38	3	29	1	0	3%	0	0

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Santa Rosa and San Jacinto Mountains Conservation Area - Palm Springs								
Le Conte's Thrasher - Other Conserved Habitat	793	103	560	554	0	99%	0	102
Peninsular Bighorn Sheep - Rec Zone 1 - Essential Habitat	9,195	226	2,511	2,220	0	88%	0	202
Desert Tortoise - Other Conserved Habitat	22,571	1,317	8,856	5,396	0	61%	0	854
Peninsular Bighorn Sheep - Rec Zone 2 - Essential Habitat	18,426	866	4,700	4,149	0	88%	0	775
Gray Vireo - Other Conserved Habitat	8,416	431	3,883	1,837	0	47%	0	227
Desert dry wash woodland	40	4	36	41	0	100%	0	5
Peninsular juniper woodland & scrub	7,682	353	3,177	1,837	0	58%	0	219
Semi-desert chaparral	733	51	571	0	0	0%	0	5
Southern sycamore-alder riparian woodland	30	2	24	0	0	0%	0	0
Sonoran cottonwood-willow riparian forest	58	0	58	4	0	7%	0	0
Desert fan palm oasis woodland	218	9	76	52	0	69%	0	6
Southern arroyo willow riparian forest	16	0	0	0	0	0%	0	0
Santa Rosa and San Jacinto Mountains Conservation Area - Rancho Mirage								
Desert Tortoise - Other Conserved Habitat	5,249	147	1,326	1,205	0	91%	0	135
Le Conte's Thrasher - Other Conserved Habitat	19	2	17	0	0	0%	0	0
Peninsular Bighorn Sheep - Rec Zone 2 - Essential Habitat	5,262	42	450	1,209	0	100%	0	106
Desert dry wash woodland	19	1	9	4	0	44%	0	1

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Santa Rosa and San Jacinto Mountains Conservation Area - Riverside County								
Peninsular Bighorn Sheep - Rec Zone 2 - Essential Habitat	14,558	647	4,269	3,032	0	71%	0	478
Le Conte's Thrasher - Other Conserved Habitat	9,123	911	5,508	5,338	0	97%	0	886
Triple-ribbed Milkvetch - Known Locations	0	0	0	0	0	0%	0	0
Peninsular Bighorn Sheep - Rec Zone 1 - Essential Habitat	24,840	830	7,252	1,267	0	17%	0	214
Gray Vireo - Other Conserved Habitat	58,985	881	7,930	6,057	15	76%	0	694
Peninsular Bighorn Sheep - Rec Zone 3 - Essential Habitat	50,972	683	5,359	5,322	78	99%	0	679
Desert Tortoise - Other Conserved Habitat	86,875	2,950	23,856	16,098	73	67%	7	2,080
Peninsular Bighorn Sheep - Rec Zone 4 - Essential Habitat	34,597	258	2,325	7,522	0	100%	0	777
Southern sycamore-alder riparian woodland	518	12	117	5	0	4%	0	2
Red shank chaparral	12,514	253	2,274	1,814	3	80%	0	207
Semi-desert chaparral	16,869	233	2,093	928	0	44%	0	116
Peninsular juniper woodland & scrub	29,547	418	2,899	3,319	12	100%	0	473
Southern arroyo willow riparian forest	16	2	15	0	0	0%	0	0
Desert dry wash woodland	3,566	298	1,244	1,272	0	100%	0	304
Desert fan palm oasis woodland	716	45	404	0	0	0%	0	5

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Snow Creek/Windy Point Conservation Area - Palm Springs								
Coachella Valley Milkvetch - Core Habitat	910	91	816	436	0	53%	0	53
Peninsular Bighorn Sheep - Essential Habitat	180	16	144	153	0	100%	0	17
Coachella Valley Round-tailed Ground Squirrel - Core Habitat	934	93	838	455	0	54%	0	55
Coachella Valley Fringe-toed Lizard - Core Habitat	749	75	672	317	0	47%	0	39
Coachella Valley Giant Sand-treader Cricket - Core Habitat	749	75	672	317	0	47%	0	39
Coachella Valley Jerusalem Cricket - Core Habitat	908	90	815	434	0	53%	0	52
Palm Springs Pocket Mouse - Core Habitat	934	93	838	455	0	54%	0	55
Le Conte's Thrasher - Other Conserved Habitat	864	86	775	404	0	52%	0	49
Ephemeral sand fields	680	68	610	266	0	44%	0	33
Active desert dunes	69	7	62	51	0	82%	0	6
Highway 111 - Whitewater River Biological Corridor	276	27	247	260	0	100%	0	28

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Snow Creek/Windy Point Conservation Area - Riverside County								
Coachella Valley Milkvetch - Core Habitat	1,700	134	1,210	766	0	63%	0	90
Coachella Valley Round-tailed Ground Squirrel - Core Habitat	1,880	152	1,371	848	0	62%	0	100
Coachella Valley Fringe-toed Lizard - Core Habitat	625	55	502	348	0	69%	0	40
Peninsular Bighorn Sheep - Essential Habitat	525	49	443	130	0	29%	0	18
Coachella Valley Giant Sand-treader Cricket - Core Habitat	625	56	501	348	0	69%	0	41
Le Conte's Thrasher - Other Conserved Habitat	1,924	162	1,453	894	0	62%	0	106
Coachella Valley Jerusalem Cricket - Core Habitat	782	60	538	518	0	96%	0	58
Ephemeral sand fields	468	45	409	348	0	85%	0	39
Stabilized shielded sand fields	157	10	93	157	0	100%	0	16
Highway 111 - Whitewater River Biological Corridor	474	46	415	145	0	35%	0	19
Stubbe and Cottonwood Canyons Conservation Area - Riverside County								
Desert Tortoise - Core Habitat	5,735	253	2,276	1,000	0	44%	29	96
Le Conte's Thrasher - Other Conserved Habitat	1,265	123	1,111	824	0	74%	0	94
Desert dry wash woodland	289	26	229	137	0	60%	0	17
Sonoran cottonwood-willow riparian forest	267	3	25	0	0	0%	0	0
Sand Transport	1,375	125	1,129	828	0	73%	0	95
Stubbe Canyon Wash Corridor	1,181	117	1,058	877	0	83%	0	99

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Thousand Palms Conservation Area - Riverside County								
Coachella Valley Round-tailed Ground Squirrel - Core Habitat	8,295	450	2,886	1,890	7	65%	39	271
Coachella Valley Milkvetch - Core Habitat	4,403	111	1,001	1,013	3	100%	5	107
Desert Pupfish - Refugia Locations	0	0	0	0	0	0%	0	0
Coachella Valley Fringe-toed Lizard - Core Habitat	3,962	93	834	685	0	82%	0	78
Le Conte's Thrasher - Other Conserved Habitat	10,539	505	3,671	1,856	7	51%	34	246
Predicted Flat-tailed Horned Lizard - Core Habitat	4,118	94	870	715	2	82%	1	78
Mecca Aster - Core Habitat	11,540	277	2,623	2,001	646	76%	5	213
Coachella Valley Giant Sand-treader Cricket - Core Habitat	3,962	93	834	685	0	82%	0	78
Palm Springs Pocket Mouse - Core Habitat	11,167	468	3,399	1,837	8	54%	38	236
Desert dry wash woodland	748	4	34	2	0	6%	0	1
Active sand fields	3,543	91	820	679	0	83%	0	77
Active desert dunes	421	2	14	6	0	43%	0	1
Desert fan palm oasis woodland	137	0	0	0	0	0%	0	0
Sonoran cottonwood-willow riparian forest	4	0	0	0	0	0%	0	0
Mesquite hummocks	58	0	0	0	0	0%	0	0
Sand Transport	12,011	519	3,615	1,796	14	50%	52	232
Sand Source	12,952	402	3,227	2,847	639	88%	5	354
Thousand Palms Linkage	24,965	919	7,238	4,623	654	64%	57	563
Thousand Palms Conservation Area - City of Indio								
Mecca Aster - Core Habitat	205	20	53	204	0	100%	0	71
Predicted Flat-tailed Horned Lizard - Core Habitat	30	3	7	0	0	0%	0	0
Le Conte's Thrasher - Other Conserved Habitat	519	47	208	371	8	100%	0	80
Coachella Valley Round-tailed Ground Squirrel - Core Habitat	218	18	88	91	8	100%	0	19
Palm Springs Pocket Mouse - Core Habitat	540	50	189	392	8	100%	0	98
Sand Transport	539	54	485	434	8	89%	0	49
Sand Source	104	10	94	104	0	100%	0	11
Thousand Palms Linkage	642	64	578	538	8	93%	0	60

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Upper Mission Creek/Big Morongo Canyon Conservation Area - Desert Hot Springs								
Coachella Valley Jerusalem Cricket - Other Conserved Habitat	49	0	49	40	0	82%	1	-1
Le Conte's Thrasher - Other Conserved Habitat	1,832	288	1,409	1,058	27	75%	2	221
Palm Springs Pocket Mouse - Core Habitat	1,748	270	1,403	1,047	27	75%	2	206
Little San Bernardino Mountains Linanthus - Core Habitat	1,020	53	967	669	16	69%	0	38
Desert Tortoise - Core Habitat	3,554	0	1,429	1,047	27	73%	0	0
Desert dry wash woodland	135	6	58	32	5	55%	0	4
Sand Transport	1,869	286	1,399	1,112	47	79%	2	231
Sand Source	343	0	6	0	0	0%	0	0
Highway 62 Corridor	73	7	66	313	0	100%	0	31
Upper Mission Creek/Big Morongo Canyon Conservation Area - Palm Springs								
Le Conte's Thrasher - Other Conserved Habitat	24	2	22	0	0	0%	0	0
Palm Springs Pocket Mouse - Other Conserved Habitat	24	2	22	0	0	0%	0	0

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Upper Mission Creek/Big Morongo Canyon Conservation Area - Riverside County								
Desert Tortoise - Core Habitat	24,122	887	7,984	5,340	40	67%	23	600
Triple-ribbed Milkvetch - Core Habitat	819	47	426	421	0	99%	0	46
Coachella Valley Jerusalem Cricket - Other Conserved Habitat	666	52	460	43	0	9%	11	-1
Le Conte's Thrasher - Other Conserved Habitat	1,871	146	1,323	632	0	48%	3	74
Palm Springs Pocket Mouse - Core Habitat	1,937	151	1,363	765	2	56%	2	89
Little San Bernardino Mountains Linanthus - Core Habitat	1,390	122	1,100	677	0	62%	0	80
Southern sycamore-alder riparian woodland	104	6	52	60	0	100%	0	7
Desert dry wash woodland	125	8	76	49	0	64%	0	5
Sonoran cottonwood-willow riparian forest	100	8	76	78	0	100%	0	8
Sand Transport	2,279	168	1,509	849	5	56%	0	102
Sand Source	19,789	721	6,488	4,733	35	73%	0	545
Highway 62 Corridor	907	79	715	276	0	39%	0	35
West Deception Canyon Conservation Area - Riverside County								
Sand Source	1,302	118	1,063	904	0	85%	0	102
Whitewater Canyon Conservation Area - Desert Hot Springs								
Desert Tortoise - Core Habitat	56	0	0	0	0	0%	0	0
Sand Source	56	0	0	0	0	0%	0	0

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Whitewater Canyon Conservation Area - Riverside County								
Desert Tortoise - Core Habitat	4,438	120	1,084	742	0	68%	1	85
Arroyo Toad - Core Habitat	2,082	78	706	676	0	96%	0	75
Little San Bernardino Mountains Linanthus - Other Conserved Habitat	579	39	348	277	0	80%	0	32
Triple-ribbed Milkvetch - Core Habitat	1,295	41	368	277	0	75%	0	32
Desert fan palm oasis woodland	1	0	0	0	0	0%	0	0
Sonoran cottonwood-willow riparian forest	166	11	107	105	0	98%	0	11
Sand Transport	1,392	48	435	338	0	78%	0	38
Sand Source	12,616	94	850	618	0	73%	1	70
Whitewater Canyon Corridor	223	22	201	0	0	0%	1	1
Whitewater Floodplain Conservation Area - Cathedral City								
Coachella Valley Milkvetch - Core Habitat	107	7	61	0	0	0%	0	1
Coachella Valley Round-tailed Ground Squirrel - Core Habitat	105	7	59	0	0	0%	0	1
Coachella Valley Fringe-toed Lizard - Core Habitat	107	7	61	0	0	0%	0	1
Le Conte's Thrasher - Other Conserved Habitat	107	7	61	0	0	0%	0	1
Palm Springs Pocket Mouse - Core Habitat	107	7	61	0	0	0%	0	1
Coachella Valley Giant Sand-treader Cricket - Core Habitat	107	7	61	0	0	0%	0	1
Active sand fields	49	5	43	0	0	0%	0	1
Whitewater River Corridor	28	2	18	0	0	0%	0	0

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Whitewater Floodplain Conservation Area - Palm Springs								
Coachella Valley Round-tailed Ground Squirrel - Core Habitat	5,825	328	2,955	865	0	29%	42	77
Coachella Valley Milkvetch - Core Habitat	5,432	297	2,671	850	0	32%	37	78
Palm Springs Pocket Mouse - Core Habitat	6,173	347	3,122	882	0	28%	61	62
Coachella Valley Fringe-toed Lizard - Core Habitat	5,418	295	2,659	846	0	32%	37	77
Coachella Valley Giant Sand-treader Cricket - Core Habitat	5,418	295	2,659	846	0	32%	37	77
Le Conte's Thrasher - Other Conserved Habitat	6,495	381	3,433	896	0	26%	61	67
Ephemeral sand fields	2,873	132	1,185	520	0	44%	10	55
Stabilized desert sand fields	577	44	394	5	0	1%	0	5
Active sand fields	436	44	392	332	0	85%	0	38
Whitewater River Corridor	1,183	90	809	50	0	6%	13	1
Whitewater Floodplain Conservation Area - Riverside County								
Coachella Valley Milkvetch - Core Habitat	96	6	58	0	0	0%	0	1
Coachella Valley Round-tailed Ground Squirrel - Core Habitat	185	11	100	0	0	0%	0	1
Coachella Valley Giant Sand-treader Cricket - Core Habitat	92	6	57	0	0	0%	0	1
Coachella Valley Fringe-toed Lizard - Core Habitat	92	6	57	0	0	0%	0	1
Palm Springs Pocket Mouse - Core Habitat	701	53	477	0	0	0%	10	-5
Le Conte's Thrasher - Other Conserved Habitat	706	53	480	0	0	0%	10	-5
Ephemeral sand fields	86	6	52	0	0	0%	0	1
Stabilized desert sand fields	5	1	4	0	0	0%	0	0
Whitewater River Corridor	701	53	475	0	0	0%	10	-5

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Willow Hole Conservation Area - Cathedral City								
Coachella Valley Round-tailed Ground Squirrel - Core Habitat	1,485	140	1,256	714	18	57%	0	86
Coachella Valley Milkvetch - Core Habitat	938	87	782	282	15	36%	0	37
Coachella Valley Fringe-toed Lizard - Core Habitat	264	24	212	156	0	74%	0	18
Palm Springs Pocket Mouse - Core Habitat	1,147	107	959	705	11	74%	0	81
Le Conte's Thrasher - Other Conserved Habitat	1,795	167	1,505	744	25	49%	0	91
Ephemeral sand fields	227	20	178	119	0	67%	0	14
Active sand fields	37	4	33	46	0	100%	0	5
Stabilized desert sand fields	57	6	51	0	0	0%	0	1
Stabilized desert dunes	1	0	1	0	0	0%	0	0
Sand Transport	966	89	798	683	7	86%	0	77
Sand Source	833	79	710	61	18	9%	0	14
Willow Hole Conservation Area - Desert Hot Springs								
Coachella Valley Milkvetch - Core Habitat	959	96	863	419	0	49%	0	52
Coachella Valley Fringe-toed Lizard - Core Habitat	3	0	3	0	0	0%	0	0
Le Conte's Thrasher - Other Conserved Habitat	1,666	167	1,499	689	0	46%	0	86
Coachella Valley Round-tailed Ground Squirrel - Core Habitat	3	0	3	0	0	0%	0	0
Palm Springs Pocket Mouse - Core Habitat	1,713	171	1,542	712	0	46%	0	88
Ephemeral sand fields	610	61	549	260		47%	0	32
Stabilized desert dunes	139	14	125	51	0	41%	0	7
Stabilized desert sand fields	54	5	49	9		18%	0	1
Mesquite hummocks	30	3	27	16	0	58%	0	2
Sand Transport	1,713	171	1,542	712	0	46%	0	88
Mission Creek / Willow Wash Biological Corridor	308	31	277	140	0	51%	0	17

	Total Acres in Conservation Area	Acres of Disturbance Authorized (1996)	Remaining Acres To Be Conserved (1996)	Acres Conserved Since 1996	Acres Conserved in 2020	Percentage of Required Conservation Acquired	Acres of Permitted Disturbance	Acres of Rough Step
Willow Hole Conservation Area - Riverside County								
Coachella Valley Fringe-toed Lizard - Core Habitat	633	50	454	313	0	69%	6	30
Coachella Valley Milkvetch - Core Habitat	2,228	195	1,751	838	0	48%	6	97
Palm Springs Pocket Mouse - Core Habitat	3,465	298	2,684	911	0	34%	6	115
Le Conte's Thrasher - Other Conserved Habitat	3,601	298	2,677	918	0	34%	6	116
Desert saltbush scrub	169	17	152	137	0	90%	0	15
Mesquite hummocks	125	11	98	76	0	77%	0	9
Desert fan palm oasis woodland	1	0	0	0	0	0%	0	0
Stabilized desert sand fields	144	14	128	62	0	48%	2	6
Stabilized desert dunes	383	35	319	149	0	47%	4	14
Ephemeral sand fields	906	81	728	101	0	14%	0	18
Sand Transport	3,500	304	2,734	910	0	33%	6	115
Sand Source	186	2	17	8	0	48%	0	1
Mission Creek / Willow Wash Biological Corridor	509	44	397	0	0	0%	0	4

Appendix IV

Covered Activity Impact Outside Conservation Areas

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Arroyo Toad	
Riverside County	0
Arroyo Toad Total	0
California Black Rail	
Coachella	0
Indio	0
Riverside County	0
California Black Rail Total	0
Coachella Valley Fringe-toed Lizard	
Cathedral City	581
Coachella	9
Indian Wells	699
Indio	999
La Quinta	556
Palm Desert	1,039
Palm Springs	1,462
Rancho Mirage	1,055
Riverside County	616
Coachella Valley Fringe-toed Lizard Total	7,016

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Coachella Valley Giant Sand-treader Cricket	
Cathedral City	581
Coachella	9
Indian Wells	699
Indio	999
La Quinta	556
Palm Desert	1,039
Palm Springs	1,462
Rancho Mirage	1,055
Riverside County	616
Coachella Valley Giant Sand-treader Cricket Total	7,016
Coachella Valley Jerusalem Cricket	
Cathedral City	591
Desert Hot Springs	10
Palm Desert	6
Palm Springs	1,468
Rancho Mirage	1,006
Riverside County	130
Coachella Valley Jerusalem Cricket Total	3,211

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Coachella Valley Milkvetch	
Cathedral City	517
Desert Hot Springs	13
Indian Wells	604
La Quinta	1
Palm Desert	1,027
Palm Springs	1,006
Rancho Mirage	1,055
Riverside County	361
Coachella Valley Milkvetch Total	4,584
Coachella Valley Round-tailed Ground Squirrel	
Cathedral City	800
Coachella	23
Desert Hot Springs	567
Indian Wells	1,028
Indio	1,531
La Quinta	1,427
Palm Desert	1,390
Palm Springs	1,444
Rancho Mirage	1,207
Riverside County	2,443
Coachella Valley Round-tailed Ground Squirrel Total	11,860

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Crissal Thrasher	
Cathedral City	0
Coachella	36
Desert Hot Springs	0
Indian Wells	21
Indio	258
La Quinta	673
Riverside County	260
Crissal Thrasher Total	1,248
Desert Pupfish	
Indian Wells	0
Desert Pupfish Total	0
Desert Tortoise	
Cathedral City	15
Coachella	0
Desert Hot Springs	532
Indian Wells	220
Indio	0
La Quinta	439
Palm Desert	470
Palm Springs	49
Rancho Mirage	179
Riverside County	609
Desert Tortoise Total	2,513

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Gray Vireo	
Palm Springs	0
Riverside County	29
Gray Vireo Total	29
Le Conte's Thrasher	
Cathedral City	957
Coachella	46
Desert Hot Springs	1,156
Indian Wells	1,287
Indio	1,570
La Quinta	1,784
Palm Desert	2,000
Palm Springs	1,414
Rancho Mirage	1,298
Riverside County	3,652
Le Conte's Thrasher Total	15,164
Least Bell's Vireo - Breeding Habitat	
Cathedral City	0
Coachella	4
Desert Hot Springs	1
Indian Wells	21
Indio	76
La Quinta	68
Palm Springs	0
Rancho Mirage	0
Riverside County	19
Least Bell's Vireo - Breeding Habitat Total	189

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Least Bell's Vireo - Migratory Habitat	
Cathedral City	5
Coachella	32
Desert Hot Springs	1
Indian Wells	188
Indio	182
La Quinta	667
Palm Desert	201
Palm Springs	7
Rancho Mirage	46
Riverside County	241
Least Bell's Vireo - Migratory Habitat Total	1,570
Little San Bernardino Mountains Linanthus	
Desert Hot Springs	0
Riverside County	0
Little San Bernardino Mountains Linanthus Total	0
Mecca Aster	
Indio	0
Riverside County	0
Mecca Aster Total	0

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Orocopia Sage	
Riverside County	6
Orocopia Sage Total	6
Palm Springs Pocket Mouse	
Cathedral City	805
Coachella	15
Desert Hot Springs	587
Indian Wells	1,048
Indio	1,419
La Quinta	1,285
Palm Desert	1,464
Palm Springs	1,496
Rancho Mirage	1,254
Riverside County	2,564
Palm Springs Pocket Mouse Total	11,937
Peninsular Bighorn Sheep	
Cathedral City	4
Indian Wells	2
La Quinta	127
Palm Desert	217
Palm Springs	5
Rancho Mirage	4
Riverside County	24
Peninsular Bighorn Sheep Total	383

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Potential Flat-tailed Horned Lizard	
Cathedral City	0
Desert Hot Springs	7
Palm Springs	377
Riverside County	29
Potential Flat-tailed Horned Lizard Total	413
Predicted Flat-tailed Horned Lizard	
Cathedral City	557
Coachella	3
Indian Wells	699
Indio	868
La Quinta	567
Palm Desert	1,039
Palm Springs	1,083
Rancho Mirage	1,042
Riverside County	1,139
Predicted Flat-tailed Horned Lizard Total	6,997
Southern Yellow Bat	
Cathedral City	0
Desert Hot Springs	1
Palm Springs	0
Rancho Mirage	0
Riverside County	0
Southern Yellow Bat Total	1

**CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation
Areas**

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Southwestern Willow Flycatcher - Breeding Habitat	
Cathedral City	0
Coachella	0
Desert Hot Springs	1
Indio	0
Palm Springs	0
Rancho Mirage	0
Riverside County	0
Southwestern Willow Flycatcher - Breeding Habitat Total	1
Southwestern Willow Flycatcher - Migratory Habitat	
Cathedral City	5
Coachella	36
Desert Hot Springs	1
Indian Wells	209
Indio	258
La Quinta	735
Palm Desert	201
Palm Springs	7
Rancho Mirage	46
Riverside County	261
Southwestern Willow Flycatcher - Migratory Habitat Total	1,758

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Summer Tanager - Breeding Habitat	
Cathedral City	0
Coachella	0
Desert Hot Springs	1
Indio	0
Palm Springs	0
Rancho Mirage	0
Riverside County	0
Summer Tanager - Breeding Habitat Total	1
Summer Tanager - Migratory Habitat	
Cathedral City	5
Coachella	36
Desert Hot Springs	1
Indian Wells	209
Indio	258
La Quinta	735
Palm Desert	201
Palm Springs	7
Rancho Mirage	46
Riverside County	261
Summer Tanager - Migratory Habitat Total	1,758

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Triple-ribbed Milkvetch	
Palm Springs	0
Riverside County	0
Triple-ribbed Milkvetch Total	0
Yellow Warbler - Breeding Habitat	
Cathedral City	0
Coachella	0
Desert Hot Springs	0
Indio	0
Palm Springs	0
Rancho Mirage	0
Riverside County	0
Yellow Warbler - Breeding Habitat Total	0
Yellow Warbler - Migratory Habitat	
Cathedral City	5
Coachella	36
Desert Hot Springs	2
Indian Wells	209
Indio	258
La Quinta	735
Palm Desert	201
Palm Springs	7
Rancho Mirage	46
Riverside County	261
Yellow Warbler - Migratory Habitat Total	1,759

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Yellow-breasted Chat - Breeding Habitat	
Cathedral City	0
Coachella	0
Desert Hot Springs	1
Indio	0
Palm Springs	0
Rancho Mirage	0
Riverside County	0
Yellow-breasted Chat - Breeding Habitat Total	1
Yellow-breasted Chat - Migratory Habitat	
Cathedral City	5
Coachella	36
Desert Hot Springs	1
Indian Wells	209
Indio	258
La Quinta	735
Palm Desert	201
Palm Springs	7
Rancho Mirage	46
Riverside County	261
Yellow-breasted Chat - Migratory Habitat Total	1,758

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Yuma Clapper Rail	
Coachella	0
Indio	0
Riverside County	0
Yuma Clapper Rail Total	0
Active desert dunes	
Palm Springs	0
Riverside County	5
Active desert dunes Total	5
Active sand fields	
Cathedral City	3
Indio	0
Riverside County	404
Active sand fields Total	407
Arrowweed scrub	
Riverside County	0
Arrowweed scrub Total	0
Chamise chaparral	
Riverside County	0
Chamise chaparral Total	0
Cismontane alkali marsh	
Riverside County	0
Cismontane alkali marsh Total	0

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Coastal and valley freshwater marsh	
Coachella	0
Indio	0
Riverside County	0
Coastal and valley freshwater marsh Total	0
Desert dry wash woodland	
Cathedral City	5
Coachella	0
Desert Hot Springs	1
Indian Wells	188
Indio	0
La Quinta	61
Palm Desert	201
Palm Springs	7
Rancho Mirage	46
Riverside County	0
Desert dry wash woodland Total	509
Desert fan palm oasis woodland	
Cathedral City	0
Desert Hot Springs	0
Palm Springs	0
Rancho Mirage	0
Riverside County	0
Desert fan palm oasis woodland Total	0

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Desert saltbush scrub	
Coachella	4
Indio	173
La Quinta	0
Riverside County	241
Desert saltbush scrub Total	418
Desert sink scrub	
Riverside County	0
Desert sink scrub Total	0
Ephemeral sand fields	
Cathedral City	2
Palm Springs	209
Riverside County	0
Ephemeral sand fields Total	211
Interior live oak chaparral	
Palm Springs	0
Riverside County	0
Interior live oak chaparral Total	0
Mesquite bosque	
Riverside County	0
Mesquite bosque Total	0

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Mesquite hummocks	
Cathedral City	0
Coachella	4
Desert Hot Springs	0
Indian Wells	21
Indio	76
La Quinta	68
Riverside County	19
Mesquite hummocks Total	188
Mojave mixed woody scrub	
Desert Hot Springs	0
Riverside County	0
Mojave mixed woody scrub Total	0
Mojavean pinyon & juniper woodland	
Riverside County	0
Mojavean pinyon & juniper woodland Total	0
Peninsular juniper woodland & scrub	
Palm Springs	0
Riverside County	0
Peninsular juniper woodland & scrub Total	0

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Red shank chaparral	
Riverside County	0
Red shank chaparral Total	0
Semi-desert chaparral	
Palm Springs	0
Riverside County	0
Semi-desert chaparral Total	0
Sonoran cottonwood-willow riparian forest	
Coachella	0
Indio	0
Palm Springs	0
Riverside County	0
Sonoran cottonwood-willow riparian forest Total	0
Sonoran creosote bush scrub	
Cathedral City	9
Coachella	1
Desert Hot Springs	46
Indian Wells	32
Indio	320
La Quinta	350
Palm Desert	259
Palm Springs	46
Rancho Mirage	133
Riverside County	882
Sonoran creosote bush scrub Total	2,078

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Sonoran mixed woody & succulent scrub	
Cathedral City	10
Desert Hot Springs	502
Indian Wells	0
Indio	1
La Quinta	27
Palm Desert	10
Palm Springs	149
Rancho Mirage	0
Riverside County	507
Sonoran mixed woody & succulent scrub Total	1,206
Southern arroyo willow riparian forest	
Palm Springs	0
Riverside County	0
Southern arroyo willow riparian forest Total	0
Southern sycamore-alder riparian woodland	
Palm Springs	0
Riverside County	0
Southern sycamore-alder riparian woodland Total	0
Cathedral City	0
Riverside County	0
Stabilized desert dunes Total	0

CVMSHCP Annual Report 2020 - Covered Activity Impact Outside Conservation Areas

Conservation Objective / Jurisdiction	Estimated Acres Disturbed Outside Conservation Areas
Stabilized desert sand fields	
Cathedral City	0
Indio	0
Palm Springs	19
Riverside County	3
Stabilized desert sand fields Total	22
Stabilized shielded sand fields	
Cathedral City	576
Coachella	9
Indian Wells	699
Indio	998
La Quinta	556
Palm Desert	945
Palm Springs	1,235
Rancho Mirage	1,055
Riverside County	203
Stabilized shielded sand fields Total	6,276

Appendix V

Aeolian Sand Species Monitoring Report

Coachella Valley Multiple Species Habitat Conservation Plan

Aeolian Sand Species Trends

2020



**Prepared by The University of California's Center for Conservation Biology
For The Coachella Valley Conservation Commission**

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Introduction

Precipitation is the primary driver of population growth in arid environments (Noy-Meir 1973; Kearney *et al.* 2018). Still, the relationship between population growth and rainfall is not linear; the seasonality, intensity, and amount of rainfall all have differential effects (Barrows *et al.* 2009). Monitoring in arid habitats must be able to partition the complex effects of rainfall from other anthropogenic effects to identify if management actions are warranted to reverse population declines. Biological monitoring elsewhere has typically focused on periodic counts of a species; results were limited to successful detection or not, or whether populations appear to be increasing or decreasing. However, even healthy populations increase and decrease over time in response to natural fluctuations of limiting resources, predator densities, and other factors. Such natural fluctuations do not necessarily warrant management intervention or indicate concerns over the design and implementation of a conservation program aimed at protecting that species. Occupancy or abundance data alone do not inform landowners and managers as to why changes are happening, and if intervention is appropriate, what if any management prescriptions might support population persistence. A different monitoring approach was necessary and was developed in parallel with the negotiations that resulted in the Coachella Valley MSHCP/NCCP (CVMSHCP-NCCP) (Barrows *et al.* 2005; Barrows and Allen 2007a; Barrows and Allen 2007b). That approach considered monitoring as a series of hypotheses using the varying intensity of drivers and stressors over time and space as independent variables. Changes in species' abundance are then the dependent, or response variable.

Habitat conservation efforts for the CVMSHCP-NCCP are coordinated by the Coachella Valley Conservation Commission (CVCC). While the aeolian sand habitats were initially a patchwork of hundreds of privately-owned parcels, current conservation landownership of that habitat includes the U.S. Fish and Wildlife Service National Wildlife Refuges, California Department of Fish and Wildlife Ecological Reserves, U.S. Bureau of Land Management, Coachella Valley Water District, Coachella Valley Association of Governments (CVAG), Coachella Valley Mountains Conservancy (a State of California conservancy), and Friends of the Desert Mountains (a private, non-profit organization). Individual conservation landowners are responsible for land management, while biological monitoring is funded and coordinated by the CVCC. Monitoring protocols are therefore applied evenly across the remaining habitat, independent of land ownership.

Coachella Valley Fringe-toed and Flat-tailed Horned Lizards

Sand dune habitats are dynamic; aeolian driven sands are continuously shifting in response to the wind, while new upwind sand additions are dependent on stochastic flood events bringing sediments out of the surrounding mountains (Barrows 1996). The aeolian sand habitat of the Coachella Valley includes four different natural community types that comprise the remnants of the valley's original aeolian sand landscape; they are defined by unique wind, sand, and vegetation characteristics (Table 1). Protection goals included maintaining sustaining populations of the Coachella Valley fringe-toed lizard within each of these community types. Flat-tailed horned lizards, while once much more wide spread, are now restricted to the stabilized sand fields and (less) active dunes of the Coachella Valley National Wildlife Refuge and California State Ecological Reserve. Monitoring goals focused on quantifying lizard densities in response to precipitation, the variation in habitat quality due to aeolian and fluvial sand dynamics, and anthropogenic stressors (Table 2) across each of the four natural communities.

We tested and rejected multiple approaches for visual counts of the lizards. Fisher et al. (2020) monitored fringe-toed lizards via a mark/recapture approach on a single 2.25 ha plot for +31 years, marking each resident lizard with a unique combination of three colored beads attached to the base of their tails (Fisher and Muth 1989). They were able to acquire both accurate annual population estimates and delineation of home ranges for resident lizards. Still, their method was time and effort intensive, typically requiring dozens of surveys per year, and so was impractical to apply to more than one or two plots; we needed a monitoring method that allowed us assess the lizard's populations on dozens of plots across the variation in habitat types spread over multiple conservation areas.

Our ultimate solution was to not count the lizards directly, but to quantify lizard densities using their tracks left in the fine aeolian sand. By using tracks, we eliminated the problem of the lizard's variable, inconsistent activity patterns – if any individual was active on a plot during or prior to the survey we could detect it by the diagnostic tracks it left behind. However, using tracks created challenges: what species had left those tracks and how many individuals were present? To determine how many lizards were represented by the tracks observed on each transect we used four criteria. First, we only surveyed on mornings after a night with strong

enough winds to clear all tracks from the previous day. Second, we followed each set of tracks to determine if it connected with tracks seen previously, and so whether a set of tracks were from a previously counted lizard or a new one. Third, we looked for interactions between lizards to again know if we were looking at one or multiple individuals. Fourth, there are considerable size differences between male and female lizards and between juveniles and adults, and those differences are mirrored in the track widths. Ensuring that the species-track identification was accurate was resolved with adequate training, and when in doubt following the tracks to the lizard that created them. Much like learning to count birds by their calls and songs, accurately identifying tracks is a learnable skill.

A benefit of the tracking method was that we could detect many more lizards, and so could reduce plot size to just 0.1 ha and still have adequate numbers of lizard sightings for robust statistical analyses. With smaller plots and smaller time and effort per plot we were able to survey 68 core plots (plots resurveyed every year) across the entire range of the lizards, with 4-6 repeated surveys per plot within a six-week survey window. We configured the 0.1 ha plots as 10m \times 100m rectangles. Those plots were then clustered (3-7 plots) within separate dunes or habitats within the same natural community type, with plot clusters > 500 m apart, (with the exception two clusters that were < 500 m apart as a result of a random placement) from an adjacent plot cluster. Placement of the initial plot within a cluster was random. Thereafter additional plots were either placed randomly or regularly to answer specific questions (such as edge effects). Non-random plot placements occurred within three clusters where we wanted to measure the effect of distance from a road/powerline that formed a habitat edge. Within a cluster we placed plots ≥ 50 m apart to avoid individual lizards overlapping adjacent plots. Fisher et al. (2020) identified home range sizes for CV fringe-toed lizard females ($\bar{X} = 505 \text{ m}^2$) and males ($\bar{X} = 662 \text{ m}^2$), which, assuming roughly circular home ranges, equate to home range diameters of 25-29 m, well below the 50 m separation between plots.

Table 1: Characteristics that distinguish the four aeolian sand natural communities found in the Coachella Valley, and that provide habitat for the Coachella Valley fringe-toed lizard.

AEOLIAN COMMUNITY CHARACTERISTICS	ACTIVE DUNES	STABILIZED SAND FIELDS	EPHEMERAL SAND FIELDS	HONEY MESQUITE DUNES
HABITAT AREA / NUMBER OF HABITAT FRAGMENTS	1370 ha / 5	400 ha / 1	1700 ha / 4	200 ha / 1
SAND	Deep, continuous, well-sorted fine sand with low silt or finer particle content	Well-sorted fine sands form discontinuous layers over layers with higher silt content.	Discontinuous patches of well-sorted fine sands, coarse sands, gravel, rocks, and boulders	Deep, well-sorted fine sand with low silt or finer particle content
SAND MOVEMENT	High mobility shifting dunes	Low mobility	Extremely high mobility	Low mobility
PERENNIAL AND ANNUAL PLANT COMPOSITION	Sparse perennial and annual cover: <i>Larrea</i> and <i>Atriplex</i>	Moderate cover of <i>Larrea</i> and <i>Atriplex</i>	Moderate cover of <i>Larrea</i> , <i>Psoralea</i> , <i>Croton</i> , and <i>Petalonyx</i>	High cover of mesquite, moderate cover of other shrubs: <i>Prosopis</i> , <i>Larrea</i> , <i>Atriplex</i> , and <i>Isocoma</i>
INVASIVE PLANT SPECIES	Low to moderate cover of <i>Brassica</i>	Moderate to high cover of <i>Brassica</i> and <i>Schismus</i>	Low to zero cover of invasive species	Moderate cover of <i>Brassica</i> and <i>Schismus</i>

Table 2. Primary stressors impacting the Coachella Valley fringe-toed lizard, their effects, and management responses for reducing those impacts

STRESSOR	SCALE	EFFECT	MANAGEMENT RESPONSE
CLIMATE CHANGE	Broad, but most severe at the eastern, hotter/drier conserved habitats	Reduced surface activity for the lizards, more severe droughts, reduced vegetation cover. Higher mortality and lower recruitment rates	Reduce impacts from other stressors
INVASIVE PLANT SPECIES	Localized, varies between sites, and between species. Most severe where there are lower sand transport rates	Sand stabilization, out-competes native annuals, reducing both plant and insect food resources for the lizards. Notably, insect abundance and diversity are reduced as Sahara mustard increases	Hand removal is the safest, but the scale of the infestations easily overwhelms staff or volunteers for large scale removal efforts. Removal efforts then need to be strategically targeted to the habitats with the greatest benefits
EDGE EFFECTS	Localized	Increased predation from greater roadrunners, American kestrels, and common ravens	Remove anthropogenic nesting sites and power lines used by the predators to hunt from.
LOSS OF GENETIC HETEROGENEITY	Broad, but most severe on the smallest habitat patches	Potential reduced adaptability to climate change and other stressors, as well as reduced. Otherwise unexplained population declines	Translocation of gravid females and/or hatchlings to increase heterogeneity. Adults do not appear to translocate as successfully.
LOSS OF ECOSYSTEM PROCESSES	Localized	Increased sand stabilization, reduced active, loose sand habitats	Keep sand corridors open. Recycle fugitive sand (sand on roads or otherwise unwanted areas) to sand corridors
OFF-ROAD VEHICLE TRESPASS	Localized	Reduced perennial vegetation cover. Increased debris dumping	Maintain fencing, increased law enforcement patrols

Population densities can vary as habitat characteristics vary, and responses to those shifting habitat qualities can become apparent at different scales (Morris 1987; Smith and Ballinger 2001). By collecting lizard densities at a plot scale (0.1 ha) that can be combined and analyzed as plot clusters provides analytic flexibility at multiple scales; these plot clusters then can be combined at the natural community or landscape scale. Our 68 core plots included replicates within the four natural communities as follows (plot clusters / total # of plots): 1) active dunes (4 / 18); 2) mesquite dunes (1 / 11); 3) ephemeral sand fields (3 / 18); and 4) stabilized sand fields (3 / 21).

Two to three people surveyed each plot, a professional biologist plus 1-2 volunteer community scientists. Surveyors slowly walked equidistant from each other along the length of the plot, noting and identifying all vertebrate tracks, which were then verified and recorded by the biologist. The addition of the community scientists significantly increased detection rates for lizards and their tracks (Barrows et al. 2016).

While population density is a useful metric, it is dependent on long-term habitat conditions. Due to a finite number of breeding adults, it can take multiple years for a population to shift from lower to higher densities, or due to multiple-year lifespans, to go from higher to lower densities. Population growth rate (γ) can prove to be a more sensitive response variable to shorter term changes in independent variables. Here population growth was calculated as $\gamma = \ln(N_{i+1}/N_i)$, where N_i is the population density in year i , and N_{i+1} is the population density the following year.

Independent variables

Although this region receives occasional isolated summer rain that can result in localized flooding, primary productivity and breeding success of the lizards are catalyzed by cool season rains (Noy-Meir 1973; Kearney *et al.* 2018). To illustrate the relationship between rainfall and the lizards' population dynamics we compared annual November-April rainfall totals from the eastern-most protected habitat, the Coachella Valley National Wildlife Refuge and California State Ecological Reserve. Rainfall data were collected on site and were found to be nearly identical to a nearby, internet accessible weather station in the city of Indio (<https://wrcc.dri.edu/cgi-bin/cliMAIN.pl?ca4259>). Rainfall levels do vary across the Coachella Valley, with an increase toward the western edge of the valley at the western limits of the

lizards' remaining habitat; however, the relative trajectories (drought, average rainfall, or relatively wet conditions) are consistent throughout the region. Using this rainfall metric to illustrate relationships between rainfall and lizard population dynamics throughout the lizards' range, while not precise for specific locations, provides the opportunity to assess how drought or wetter conditions influence the lizards' population densities. Rainfall levels provide a coarse-scale expectation of population growth rate trajectories.

Additional independent data that we collected annually on each 0.1 ha plot included: 1) spring annual and perennial plant abundance and density by species, including both native and non-native species; 2) arthropod abundance and species diversity, 3) sand compaction, and 4) associated vertebrates, using track counts collected at the same time that the lizards were surveyed. These metrics provided fine-scale, plot-specific indicators of habitat characteristics. For annual vegetation cover we measured both density and % cover, by species, on 12, 1m² sub-plots, four at each end and in the center of each 0.1 ha plot. We measured arthropods using three pitfall traps placed overnight, one at each end and one in the center of each 0.1 ha plot. One of those arthropods, the beetle *Asbolus* (previously *Cryptoglossa*) *laevis*, (Tenebrionidae) proved to be a useful indicator of sand compaction, only occurring on the less compacted sands of active dunes (Barrows 2000). Sand compaction was measured using a Pocket Penetrometer (AMS Inc.). Twenty-five compaction measurements, each separated by roughly 4 m, were made along the mid-line of each plot. Associated vertebrates were measured using the same track protocol used to measure the lizard densities. Some of the associated vertebrates are predators and so could influence fringe-toed lizard abundance. Potential predators include leopard lizards (*Gambelia wislizenii*), sidewinders (*Crotalus cerastes*), coachwhips (*Masticophis flagellum*), glossy snakes (*Arizona elegans*), greater roadrunners (*Geococcyx californianus*), loggerheaded shrikes (*Lanius ludovicianus*), common ravens (*Corvus corax*), American kestrels (*Falco sparverius*), coyotes (*Canis latrans*), and potentially some species of rodents (Timberlake and Washburne 1989). Roadrunner, kestrel, and raven densities increased with proximity to human development; both the roadrunner (except on the mesquite dune natural community) and kestrel were dependent on planted non-native trees and shrubs for nesting sites. Others are possible competitors such as zebra-tailed lizards (*Callisaurus draconoides*) and flat-tailed horned lizards (*Phrynosoma mcallii*), but none are as habitat specific to active aeolian sand as are fringe-toed lizards.

Coachella Valley Fringe-toed Lizards

Figure 1 illustrates the nested-scale character of the fringe-toed lizard monitoring data. At the finest scale are individual plots clustered within a single active dune (AD2). Means for the combined plots within each of the four individual active dune plot clusters (replicates within the active dune natural community) are shown in the middle scale. Finally, at the coarsest scale are the combined means for each of the four natural communities across the lizards' entire range. At each of these scales the data can reveal patterns that provide insights regarding the status of the lizard. At both the fine-scale plot level for the AD2/active dune cluster and the combined active dune natural community scale, precipitation levels positively correlate with lizard densities (Pearson's Correlation: AD2 plot cluster: d.f.= 17, $r = 0.717$, $P = 0.0008$; all active dune communities: d.f. = 17, $r = 0.581$, $P = 0.011$). At the coarsest natural community scale (Figure 3c), the correlation (r) between lizard density and precipitation was uneven. The strongest correlation was with active dunes. Next was the mesquite dunes (d.f. = 17, $r = 0.514$, $P = 0.029$), followed by non-significant rainfall-lizard density correlations for stabilized sand fields (d.f. = 17, $r = 0.317$, $P = 0.199$), and ephemeral sand fields (d.f. = 14, $r = 0.077$, $P = 0.785$).

Since the plots are replicate surveys within each dune, and the dunes are replicates within the natural community, the general within year synchrony provides validation for the ability of the plot size and survey methodology to detect real change when it happens. Large population swings are a regular occurrence and should not catalyze management responses if they are synchronized in direction and amplitude with shifting rainfall levels. The question then is when does asynchronous, or non-significant correlations between precipitation and lizard densities indicate a need for management intervention?

A list of potential stressors that could warrant management responses is shown in Table 2. Of those that have localized impacts, off-road vehicles could be discounted as no recent vehicle trespasses were observed. Invasive species impacts and losses of ecosystem processes (reduced sand delivery) can be interrelated and so are difficult to partition. However, looking at that middle scale graph, in 2020 there were opposite population trajectories for the AD2 and ADM plot clusters (increasing) versus the AD4 and ADJ clusters (decreasing). Those divergent trajectories warranted further analyses. The AD2 and ADM plot clusters did have significantly less Sahara mustard, *Brassica tournefortii*, than the AD4 and ADJ sites (Means 13.23 versus

24.75 plants/m²; ANOVA d.f. = 1, $F = 4.5313$, $P = 0.049$), and had a significantly higher (and positive) population growth rate (means $\gamma = 0.103$ versus -0.644 ; ANOVA d.f. = 1, $F = 18.9855$, $P = 0.00049$). While lizard population growth on AD2 and ADM were less than that for the Ephemeral Sand Field natural community (Figure 1), a habitat that lacked Sahara mustard, their respective population growth rates were not significantly different (means $\gamma = 0.103$ versus 0.57 ; ANOVA d.f. = 1, $F = 4.0887$, $P = 0.0561$). The mustard densities on AD4 and ADJ appear to have exceeded a tipping point for negatively impacting the lizards. An illustration of the varying Sahara mustard densities that can occur across the active dunes and stabilized sand fields are shown in Figure 4.

The regression of 2020 lizard density versus sand compaction was significant for both active dunes ($R^2 = 0.5939$; $P < 0.00001$) and stabilized sand fields ($R^2 = 0.2101$; $P < 0.003$); less compacted sand in correlated with higher densities of fringe-toed lizards (Figure 2). There appears to be a sand compaction level of approximately 0.125 kg / cm^2 that distinguishes most active dunes from stabilized sand fields. Of the AD2 and ADM plots designated *a priori* as active dunes, 75% had sand compaction levels fitting to that natural community. However, for the AD4 and ADJ active dune plots, just 30% had sand compaction levels $\leq 0.125 \text{ kg / cm}^2$. The occurrence of plots previously identified as active dunes, but now with sand compaction and lizard densities well within the stabilized sand field range, identified a need to initiate remedial management. We did not find any support for other additional explanations, such as edge effects which are manifested by increases in potentially anthropogenically augmented predator densities (i.e. roadrunners, ravens, or kestrels). Our data identified that management intervention to remove mustard as well as remove any other barriers to aeolian sand movement was warranted on the AD4 and ADJ dunes. The lack of synchrony between lizard density and coarse scale precipitation data identified that a potential problem existed; finer scale invasive species densities and sand compaction data identified the cause and management solutions.

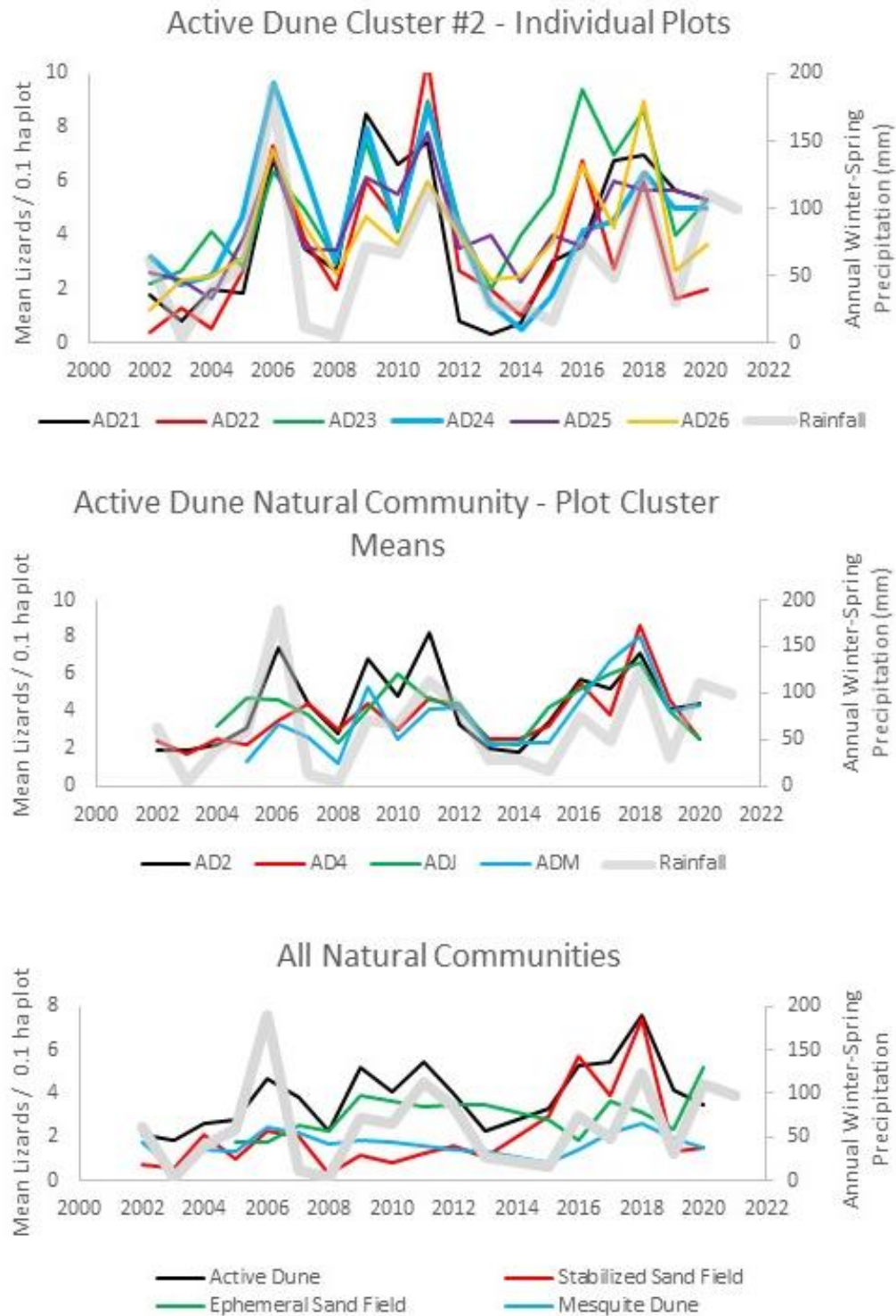


Figure 1. Annual changes in lizard density at multiple scales within the context of precipitation to show how the lizards' population fluctuations are often synchronized with rainfall patterns. Since lizard density is in part a reflection of the previous year's reproductive recruitment,

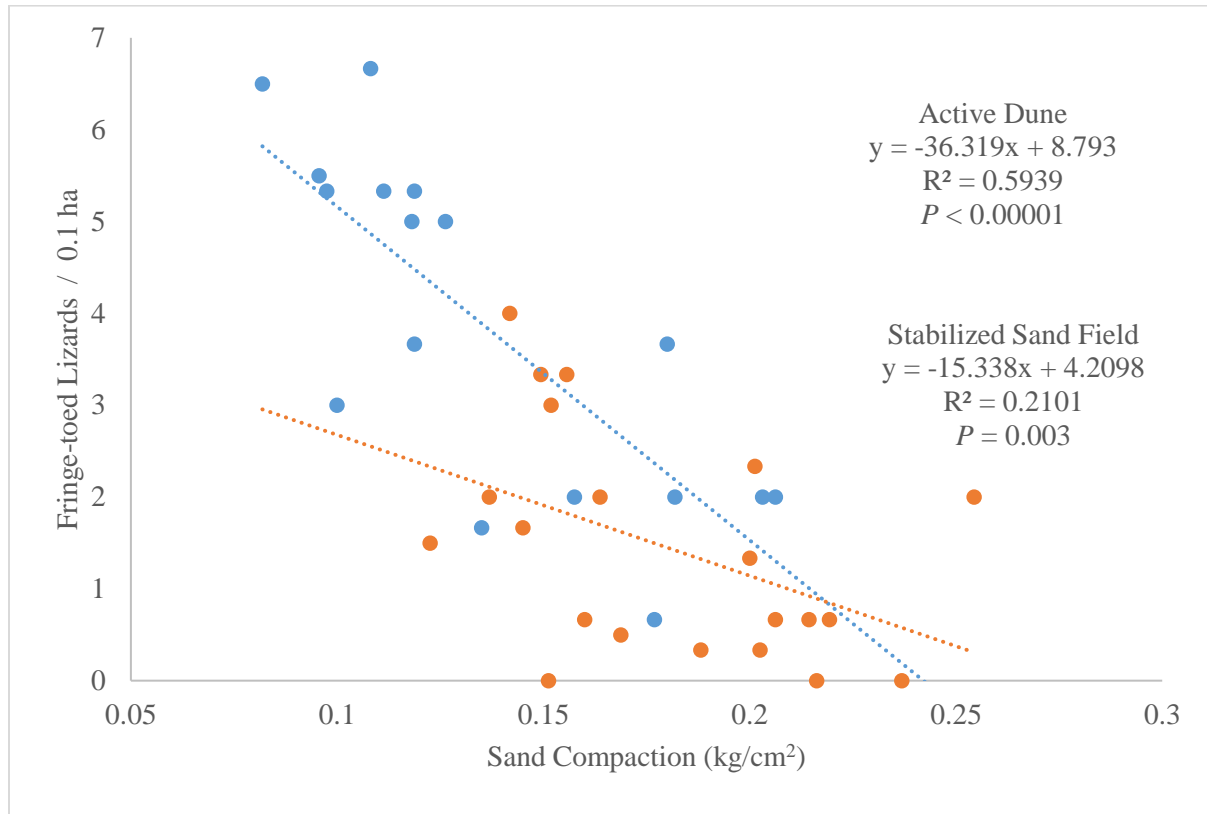


Figure 2. Patterns of Fringe-toed lizard densities in relationship to sand compaction in 2020. Active dunes (each plot indicated by a blue circle) generally have less compacted sand and higher lizard densities, whereas stabilized sand field plots (orange circles) have more compacted sand and fewer lizards. The regression of lizard density versus sand compaction for each habitat type show statistically significant correlations. The plots identified as active dunes, but that have values that are well within those for stabilized sand fields are not receiving new sand and are being invaded by Sahara mustard.

Flat-tailed Horned Lizards

In 2006 we documented an “edge effect” that was keeping flat-tailed horned lizards from being able to occupy the full extent of the habitat that has been set aside for them on the Coachella Valley National Wildlife Refuge and California State Ecological Reserve (Barrows et. al. 2006). What we found was that lizard predators, subsidized by the nesting structures unwittingly being provided for them on adjacent, non-conservation lands, had created a resident population of those predators (American kestrels and greater roadrunners) that were then utilizing the flat-tails as food to feed their chicks. Flat-tailed horned lizards typically occupy treeless aeolian sand and silty habitats where, due to the lack of nesting habitat, such avian lizard predators are absent. The lizard’s survival adaptations do not include mechanisms for avoiding such predators, especially when predators such as the kestrels can scan their hunting grounds from power lines, and then swoop down to capture the horned lizards.

While no management efforts have been employed to reduce the impact of subsidized predators on the flat-tailed horned lizards, now 14 years later, we wanted to document whether that predation-based edge effect was still occurring. Figure 3 clearly shows that it is. What

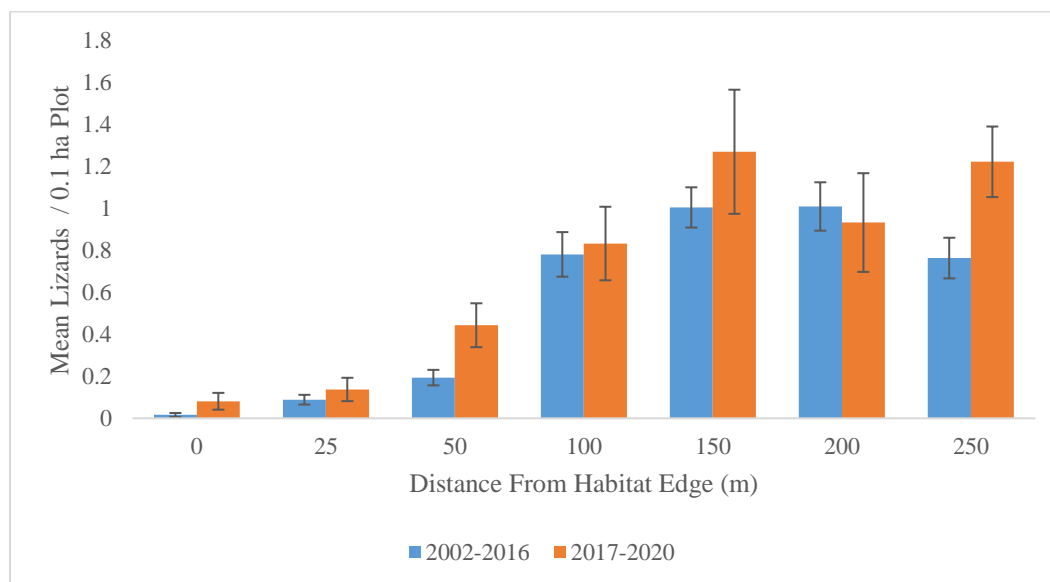


Figure 3. Densities of flat-tailed horned lizard as a function of distance from the habitat edge. Data are based on the L & H plot clusters; the temporal categories are arbitrary, aiming only to separate recent versus older population densities and to have sufficient data in each category to allow for statistical testing. Error bars indicate one standard error.

we found was that yes, the edge effect still exists. However, there has been an incremental, statistically significant increase in flat-tail densities on the 50 m from the habitat edge plots (ANOVA, d.f. = 1, $F = 7.9526$, $P = 0.0078$). There was also an increase in horned lizard density on the 250 m plots (ANOVA, d.f. = 1, $F = 5.1954$, $P = 0.02867$), perhaps reflecting an overall increase in the flat-tail populations.

As with the fringe-toed lizards, one of our aims is to identify metrics that then indicate expected temporal and spatial population fluctuations, and so when the lizard densities deviate from those expectations, management actions may be warranted. Since there is a clear edge effect impacting flat-tailed horned lizards, for understanding what those broader habitat metrics are, we have excluded data collected from plots < 100 m from habitat edges from analyses of those broader habitat constraints, as lizard densities there are predominantly influenced by edge effects and would therefore not be as sensitive to habitat features affecting lizards occupying habitats further distant from the habitat edges. Although sand compaction is an important spatial metric for identifying expected densities of fringe-toed lizards, within the range of compaction values these horned lizards have available to them, that metric has no explanatory value for flat-tailed horned lizards (Figure 4).

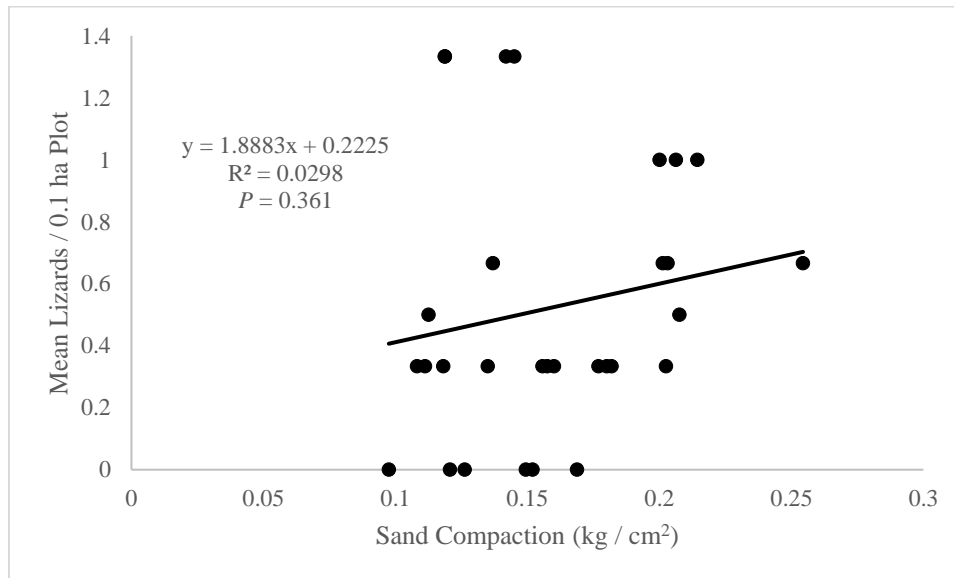


Figure 4. The relationship (or better stated the lack of a relationship) between sand compaction and 2020 flat-tailed horned lizard densities on the AD2, AD4, J, L, H, and MH plot clusters (30 plots). By looking at a single year we remove any temporal component and look exclusively and the spatial influence of sand compaction.

Temporal patterns of annual rainfall also appear to have little influence on the differences in this species' density (Figure 5). This lack of a relationship is puzzling. But may be explained by the abundance of Sahara mustard the periodically becomes extremely dense across the flat-tail's habitat. This was true for the period of 2008-2011 (see annual plant monitoring section), a time span with relatively high rainfall, but corresponding to very low horned lizard densities. The correlation between mustard cover and horned lizard densities is negative and statistically significant ($r = -0.509$, $P = 0.0373$; ANOVA, d. f. = 1, $F = 7.71$, $P = 0.009$); higher Sahara mustard densities are associated with fewer horned lizards.

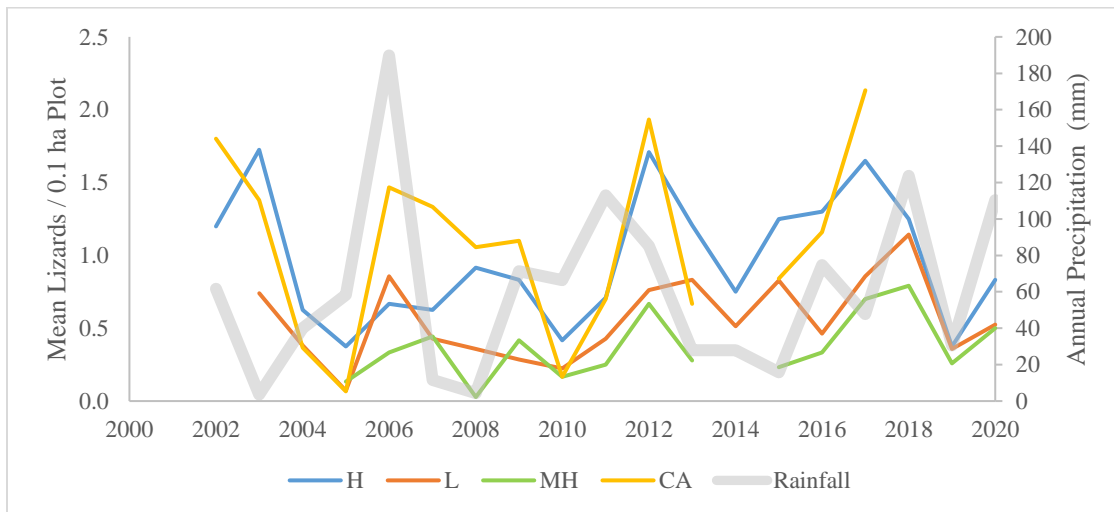


Figure 5. The relationship between annual rainfall and flat-tailed horned lizard densities on the L, H, CA, and MH plot clusters (19 plots).

The primary food for flat-tails, like all horned lizards, are ants, primarily species of harvester and honeypot ants. As such we would expect to see a relationship between those ants' abundance and the abundance of flat-tails. Figure 6 shows these temporal changes in ants and flat-tails on three of the plot clusters: H, L and MH. There are a several insights we can collect from these patterns. First, there does not appear to be a broad spatial congruence in temporal ant abundance nor flat-tail abundance, rather each plot cluster seems to be fluctuating independently. Second, early, from 2002-2007 there appears to be synchronous fluctuations with ants and flat-tails, as we hypothesized. However, from 2008-2020, that relationship evaporates into

asynchrony. We can only speculate why this apparent shift occurred. The onset of the shift was coincident with the dense Sahara mustard cover from 2008-2011, but if related, the mechanism and the reason for the on-going asynchrony even though the mustard has fluctuated since then, remains elusive. We plan on doing more extensive ant / pitfall surveys in 2021 to determine if the apparent asynchrony is the result of insufficient ant sampling.

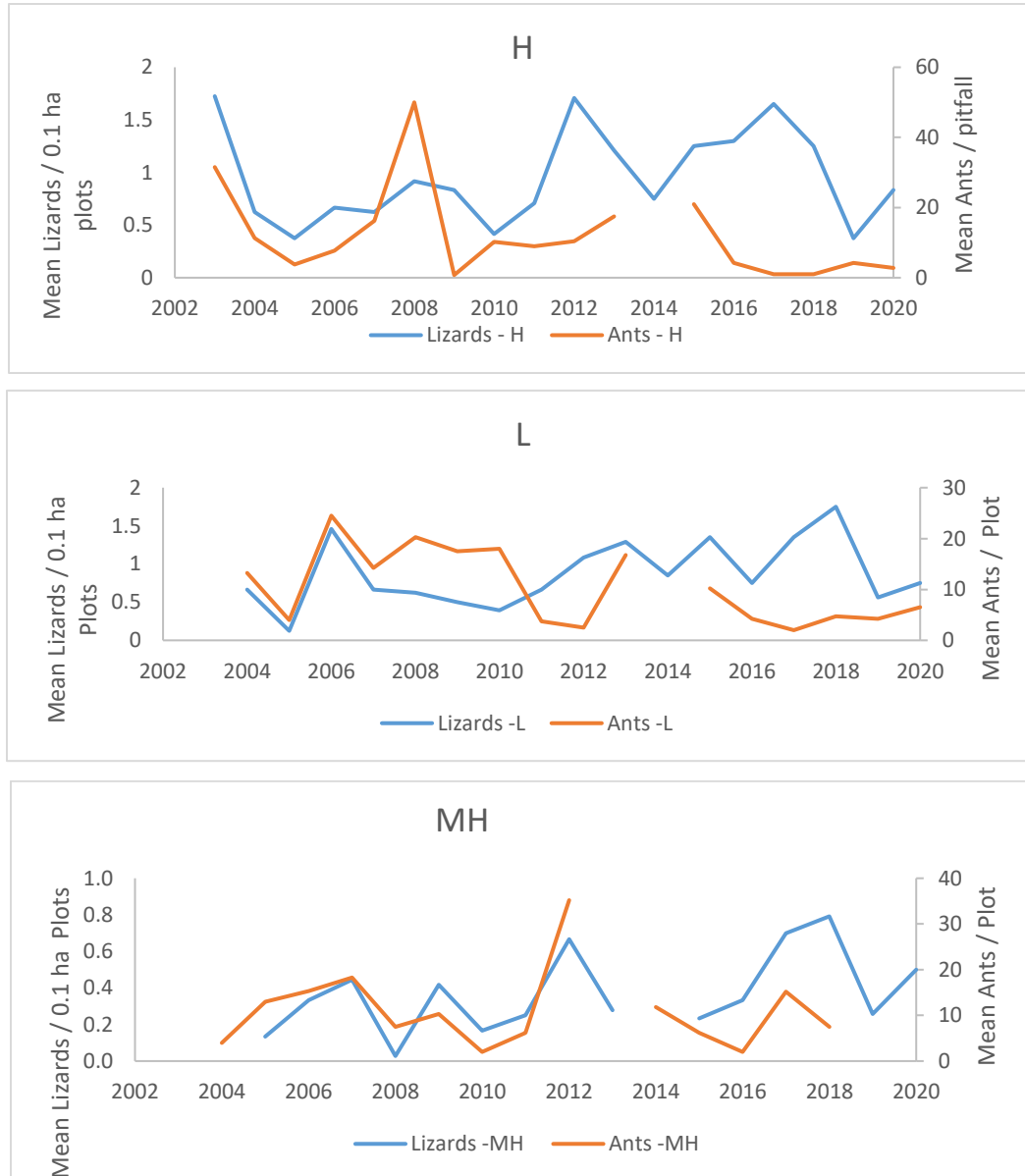


Figure 6. The relationship between ant abundance and flat-tailed horned lizard densities on the L, H, and MH plot clusters (30 plots). Ant abundance was based on the combined mean pitfall captures per plot for *Veromessor perganderi*, *Pogonomyrmex* spp, and *Myrmecosytus* spp. Ants.

DISCUSSION

Wild populations fluctuate naturally in size from year to year. The challenge for managing endangered species that are facing multiple stressors is distinguishing natural population oscillations from population shifts that are anthropogenic-driven and that, if not managed, could result in population declines leading to extinction. Here we provided examples of how the hypothesis-driven monitoring approach employed for the Coachella Valley fringe-toed lizard and flat-tailed horned lizard has clarified those distinctions and identified site-specific management recommendations. Using two abiotic metrics, precipitation (coarse scale) and sand compaction (fine scale), plus a biotic metric (invasive plant densities), we identified site-specific priorities for managing an invasive weed, Sahara mustard, to promote more sustainable lizard populations. Without management intervention, some active dune communities, habitats that where fringe-toed lizard populations are consistently the densest throughout its range, appear to be transitioning to stabilized sand fields, a natural community with consistently the lowest lizard densities. Similarly, flat-tailed horned lizards are consistently at low densities when and where the mustard is densest.

We continue to find that the mustard decreases native plant abundance (Barrows et al. 2009), decreases arthropod abundance (Hulton et al. 2013), and increases sand compaction. The result was that as Sahara mustard increased the lizards became increasingly scarce, and ultimately absent. Our findings indicate that the mustard continues to be a significant threat to the sustainability of the lizard populations, especially on stabilized sand fields and active dunes. This is in contrast to our findings that another invasive weed, Russian thistle, *Salsola tragus*, had a benign to positive impact on the lizards (Barrows 1997).

The density of the mustard is tied to both the amount of rainfall and sand transport rates, the more rainfall and the more stable the sand, the denser the mustard. It is not just the amount of rainfall, but also the timing. Heavy early December rains guarantee a dense growth of mustard, but if the rains do not start until late February or March, little mustard germinates. If there is a sequence of storms beginning in December and continuing through February, a new cohort of mustard germinates after each storm. These patterns complicate control efforts. Herbicides that kill mustard will also kill native annual plant species; following an herbicide treatment, if more storms occur, then more mustard will still germinate. That leaves “surgical” hand pulling, focusing on areas where mustard removal will yield the greatest benefits, as the primary control

method. Unless a safe, species-specific biological control for the mustard is identified, hand pulling will be an ongoing management task. Stabilized sand fields have the highest levels of Sahara mustard infestation as well as the highest sand compaction levels of any of the aeolian sand communities. The beetle *Asbolis laevis* does not occur there, and the dominance of the mustard has so far overwhelmed any effort to control it there.

Ephemeral sand fields also did not have significant correlations with precipitation; this community occurs in a region of the Coachella Valley where wind and sand transport are so strong as to continue to blow deposited sand downwind and scour rocks into ventifacts (Table 1). Within the ephemeral sand fields, due to these strong winds, sand residence time is relatively short compared to the other aeolian sand-based natural communities. These scouring winds also inhibit annual plant growth (including non-native invasive species), so higher annual rainfall that supports annual plant growth and arthropod prey for the lizards elsewhere has less of an impact on the lizard's population dynamics here. A close correlation between annual precipitation and the lizard's population growth should not be expected. Rather, when sand delivery is sufficient to build sand hummocks, and when that coincides with sequential years of average or greater rainfall to maintain high soil moisture to support leaf and flower production of perennial shrubs, the lizard population there does grow, as it did in 2020. Understanding site-specific interactions between abiotic inputs and biotic responses is critical for developing models from which the need for management interventions can be determined. For this natural community there are up-wind sand corridor challenges, such as sand and gravel mining, channelization for aquifer re-charging, and conflicts associated with roadways that cross the sand corridor. Each of these could restrict sand delivery to this habitat, and each needs to be watched to ensure sand delivery is not constrained.

We have previously addressed questions that included whether the high degree of habitat fragmentation had resulted in a loss of genetic diversity in the lizards. Based on tissue samples collected in the mid-1990s, Hedtke et al. (2007) found no genetic structure associated with the lizard populations occupying the different fragments; their genetic profile reflected the pre-fragmentation, panmictic condition. A follow-up study analyzing tissues collected in 2008, (Vandergast et al. 2016) found a different result; lizard populations occupying each habitat fragment had a unique genetic signature, and each population had lost genetic diversity relative to that 1990s baseline. Climate change also looms as a threat to the lizards. Barrows et al. (2010)

modeled the response of the fringe-toed lizards to expected levels of climate change if no significant reductions in anthropogenic greenhouse gases occur and found that only the westernmost habitat areas will likely continue to provide the climate envelope currently preferred by the lizards. Of course, models are just hypotheses in need of empirical testing, and so far, on all the remaining protected habitats the lizards are sustaining populations as expected with respect to annual rainfall and Sahara mustard densities. Given that land managers do not have the capacity to alter the course of climate change, it is imperative that they address those threats that they can affect. These include controlling invasive plants and keeping sand corridors unobstructed, and reducing other stressors that might, together with climate change, result in local extirpations.

Forty years after the listing of the Coachella Valley fringe-toed lizard as endangered, this species continues to thrive across much of the same landscape they occupied in 1980. Land protection efforts, purchasing essential private parcels and so taking them out of a trajectory toward future development, has been extremely successful. However, long-term success, defined as maintaining sustaining fringe-toed lizard populations across those protected lands, will depend on effective management informed by hypothesis-based monitoring.

Coachella Valley Milkvetch



Coachella Valley milkvetch (*Astragalus lentiginosus* var *coachellae*), is federally endangered and endemic to the Coachella Valley. It is found only in areas with abundant loose sand, as it is thought that its seeds require sand scarification to germinate. It is found at its highest density on the ephemeral sand fields of the Whitewater Floodplain Preserve, but can also be found as far east as the Coachella Valley National Wildlife Refuge and as far west as our Tipton Road plots near Windy Point. This plant is normally an annual, but with sufficient conditions it can survive multiple years; one robust specimen in Desert Hot Springs has been alive for at least three years. Our surveys of this plant consist of a simple count of all individuals present at each of our aeolian community plots.

Figure 7 shows CV milkvetch densities for 2019 and 2020. The densities between years have remained stable, except ESF 7-12, near Gene Autry Drive on the Whitewater Floodplain Preserve. In 2019, one of the plots here contained a roughly 25 m² patch supporting approximately 500 milkvetch seedlings, which accounts for most of the plants within this plot cluster. This is an unusual occurrence and accounts for the extremely high density here in 2019.

However, we recounted the milkvetch here again in late-May during the vertebrate surveys and recorded a substantially higher 22.8 mean milkvetch per 0.1ha. These new plants were a second cohort that sprouted after recent rains. It is likely these plants will not survive to maturity because they will have to contend with the harsh summer heat and aridity.

Figure 8 shows CV milkvetch densities at the CVNWR (a) and our remaining plots (b) along with SPI since 2008. CV milkvetch population density usually does not track expected patterns in response to rainfall, except for the intense drought years of 2013 and 2015 at the CVNWR where milkvetch populations were essentially undetectable. An explanation for this phenomenon is that abundant rainfall may detrimental to milkvetch germination. Since these plants require seed scarification by blowing sand, frequent rainfall that keeps the sand moist and compacted will inhibit it from blowing.

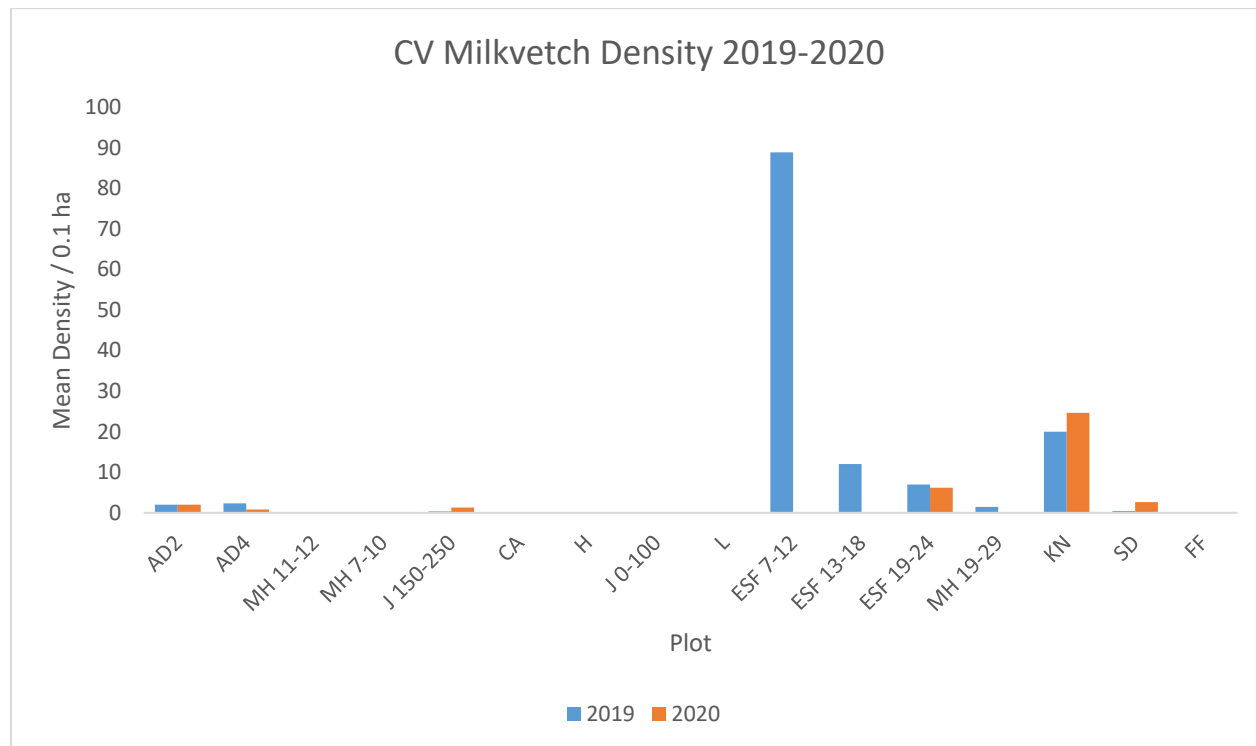


Figure 7: CV milkvetch densities for 2019 and 2020.

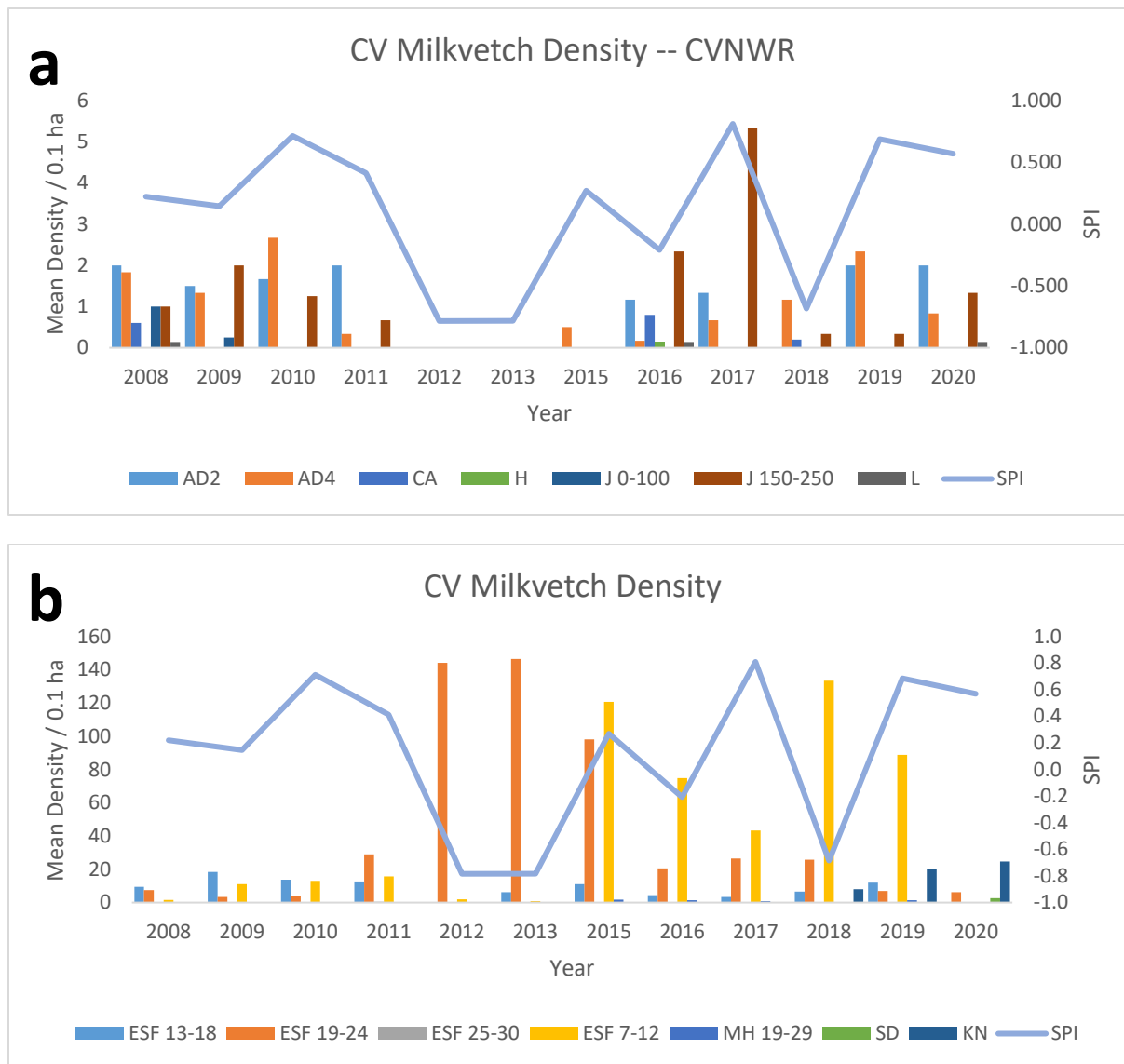


Figure 8: a.) CV milkvetch density at the CVNWR since 2008, with SPI; b.) CV milkvetch density at remaining aeolian community sites. We excluded our MH 7-10 (CVNWR) and Fingal's Finger plots because no plants have been recorded here since 2008.

Coachella Valley Giant Sand-Treader Cricket



The Coachella Valley Giant Sand-Treader Cricket (*Macrobaenetes valgum*, or CVGST) is a large wingless camel cricket of the family Rhaphidophoridae. It is a protected species under the CVMSHCP, as it is only found in areas with large amounts of fine, active sand, such as sand dunes and mesquite hummocks, which have drastically declined in area as a result of development and blocking/alteration of sand sources. However, at areas with remaining healthy dunes, such as the CVNWR, these crickets can be among the most abundant large arthropods. Little is known of their biology, but their lifecycles appear to be closely linked to late summer or winter rains. Nymphs (juveniles) are present in large numbers in the autumn months but are too small to be easily detectable. However, the crickets grow rapidly throughout winter, and by late winter or early spring the surviving CVGST are large enough to survey. CVGST are nocturnal foragers that likely feed opportunistically on plant and animal matter. They are sensitive to high heat, so each morning, before the heat of the day arrives, they excavate a new burrow into the sand presumably to a depth where conditions are comfortably cool and moist. Their method of excavation leaves behind a characteristic triangle-shaped pile of sand tailings at the mouth of

each burrow. By July or August, when summer temperatures are at their maximum, adult CVGST have mostly disappeared.

Methods

Our surveys of CVGST take place in late winter to early spring, when the crickets are large enough and abundant enough to detect. We conduct surveys across all our 0.1ha aeolian community plots. This year, we conducted surveys from 13 February to 15 April. Each plot is surveyed once during the monitoring season. CVGST are recorded by counting the diagnostic triangle-shaped sand piles at the mouths of their burrows. This method is useful for a variety of reasons. These sand piles are distinctive enough as to be rarely confused with a burrow from another species, even to a novice observer. Secondly, counting burrows is an activity that can be conducted by our team during daylight hours. Otherwise, we would either have to monitor the crickets at night when they are above-ground and detectable, or we would have to excavate them from the ground during the day, which would seriously risk the health of the animal. Only burrows that are “closed” (the entrance is blocked with sand) are recorded, as this indicates that a CVGST is occupying the burrow. Additionally, because crickets may burrow into similar areas each morning, only one cricket is counted per square meter in order to avoid duplicate records, unless the tailings are of notably different size which indicates two separate crickets are occupying the same area.

Results

CVGST are thought to rely heavily on winter rains to complete their lifecycle. It has been observed in the past that CVGST density generally follows changes in winter rain. 2018 was a drought year, but 2019 and 2020 were relatively wet years. As shown in Figure CVGST 1(a-c), CVGST densities in 2019 were comparable to 2018, so we hypothesized that there was a lag time in the population rebound, and given sufficient winter rainfall, higher CVGST densities would be seen in 2020. However, the opposite appears to be true – across all monitoring sites except for Tipton Road, Stebbin’s Dune, and the Kim Nicol Trail, CVGST densities are lower than in 2019, and in many cases they are also lower than the drought year of 2018. The cause of this pattern is not known; however, a few possibilities exist. First, continued stabilization of dunes may be reducing available loose sand habitat. Similarly, increasing Sahara mustard density may be altering available habitat and food sources for the crickets. Also, the decrease in CVGST density may be correlated to an increase in predators, especially small mammals such as

kangaroo rats. At the CA plot cluster, almost all the sand was washed away during flooding in the winter of 2019, which accounts for the decrease in crickets here.

The overall observed downward trend is likely due to complex interactions involving a multitude of variables, but an increase of predators is perhaps one of the more important factors. Most predatory lizards that share the same range as CVGST are diurnal, so rate of predation from these is likely low. Nocturnal reptiles, such as geckoes and various snakes, as well as predatory arthropods, such as large arachnids, undoubtedly take CVGST when possible, but no evidence exists that these predators have experienced an increase in abundance. However, through our aeolian community vertebrate tracking efforts, we have demonstrated that small mammals (pocket mice and kangaroo rats) have greatly increased in abundance in 2020. While largely seed-gatherers, these rodents, especially kangaroo rats, are known to consume arthropods when presented the opportunity. An increase in rodent abundance as a result of relatively wet winters in 2019 and 2020, coupled with the possibility of a decrease in adequate plant food sources due to the dominance of Sahara mustard, may have led to a notable increase in predation on arthropods, including CVGST.

In 2019 we hypothesized that roadside clearing of blowsand along Avenue 38 bordering the CVNWR may actually be beneficial for CVGST due to the creation of sand berms which serve as artificial dunes. Our regularly surveyed plot clusters that border Avenue 38 are H, J, and L. Plots at the “000” mark (0 meters away from the road) partly include the graded shoulder of Avenue 38. For L and J, and especially in 2019, strong evidence exists that the CVGST were benefitting from roadside grading, as shown by dramatically increased density on the 000 plots demonstrated in Figure CVGST 1(d-f). However, in 2020, this does not appear to be the case – centers of density are shifted more toward the midway plots (roughly around the 100m plots), or in the case of H, crickets are largely absent from all plots except H 250. H has historically had the lowest cricket density out of these three roadside plots, so these results are not surprising given the hypothesized overall downward trend. The absence of crickets at the 000 plots at J and L may be due to decreased or altered grading activity along Avenue 38, leading to stabilization of these berms. We did not track grading activity in this area, however.

Tracking grading schedules and intensities on Avenue 38 will help determine if increased grading causes an increase in CVGST density on the resulting sand berms, and if a reduction in grading likewise causes a reduction in CVGST density due to stabilization of the berms.

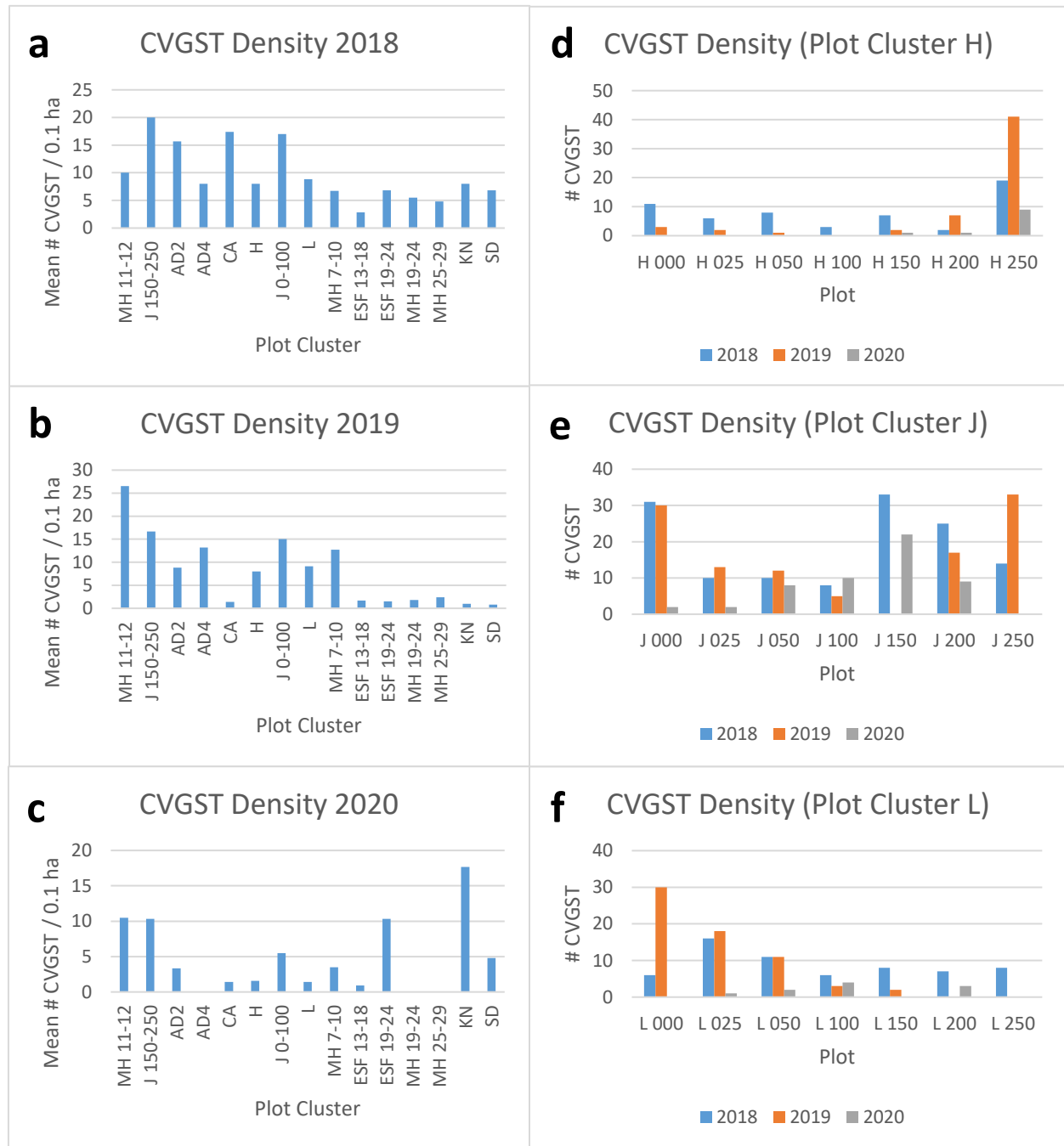


Figure 9. CVGST density across our study plots for 2018, 2019, and 2020. Fig1 a-c: CVGST density by plot cluster. Plots are plotted roughly east to west (left to right, respectively). Fig 1 d-f: CVGST abundance per plot by year for plot clusters H, J, and L on the CVNWR.

Annual Plant Monitoring

We surveyed native and invasive annual plant abundance and coverage 1m x 1m quadrats arranged along our 0.1 ha plots (Figure 6). Following an extremely successful year for native annuals on the CVP in 2017, overall percent cover of both native and invasive annuals has predictably returned to historically low levels due to lower winter precipitation this year (Figure 1). The high coverage of invasive annuals from 2008 to 2011, particularly on the CVP (AD and SSF), was mostly comprised of Sahara Mustard and is a result of consecutively early winter rains which this plant favors (Figure 7). In contrast, the high coverage of annuals on the CVP in 2017 was the result of above-average amounts of late winter precipitation, which the native plants favor. A combination of drier conditions and later winter rains since 2012 has resulted in an overall reduced coverage of invasive annuals.

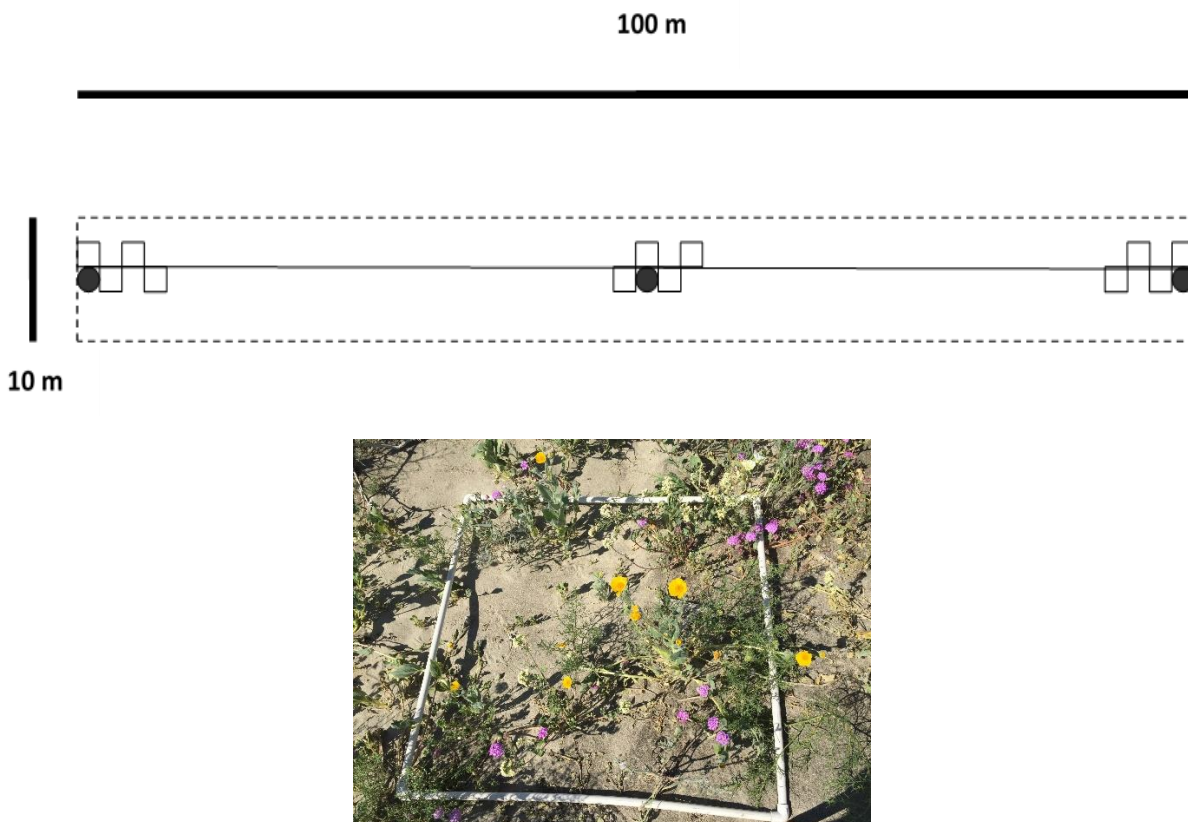


Figure 10. Schematic of plot design. The twelve small squares show the layout of the m² frames where annual vegetation density and cover is measured. The three solid circles represent where arthropod pitfalls are placed. The center lined running the length of the plot is used as a line intercept to quantify perennial plant cover on the plot.

Our yearly vegetation surveys took place from February 13 to April 15. These surveys utilize twelve 1m x 1m quadrats placed regularly within each of our 10m x 100m aeolian community plots (Figure 10) to estimate annual plant abundance, species richness, and cover. We also record seedling perennial plants in this survey because they are functionally the same as annual plants. Both native and invasive/exotic annuals are recorded.

For all plots on the CVNWR, both annual plant density (Figure 11) and Sahara mustard density (Figure 11-12) have either remained stable, or, more commonly, increased since 2019. This increase is in spite of slightly lower winter SPI versus 2019 and is likely due to a replenished seed bank after the 2019 “super bloom”. We did not observe the same drastic increase in plant density across our remaining aeolian community plots at the Whitewater Floodplain Preserve, Willow Hole, Stebbins Dune, Kim Nicol Trail, Windy Point, and Fingal’s Finger. Deep, loose sand, such as on the CVNWR, typically retains soil moisture for longer and may be a primary reason for the higher plant densities observed there. Also, the ephemeral sand fields of the Whitewater Floodplain Preserve experience extremely high winds which reduces the density of plants growing there.

In the past, nonnative plants have been shown to exert a negative effect on annual plants. This is clearly apparent during the periods of 2008 to 2011 (high invasive density/cover and low native density/cover) and 2016-2019 (low invasive density/cover and higher native density/cover). The dominant invasive plant, especially on the CVNWR, is Sahara mustard (*Brassica tournefortii*). Mediterranean grass (*Schismus barbatus*) can also reach high densities, but rarely obtains high coverage and so likely poses much less competition for native annuals. Interestingly, this year we observed little difference between invasive and native plant abundance and cover at the CVNWR. This may be due to the mustard reaching full size at a later time than usual – field observations taken in May and June at the CVNWR indicate that Sahara mustard has become the dominant plant. Also, winter rains started later in the season than in 2019, which is thought to be less ideal for mustard growth, and hence may have given the native annuals enough of an advantage to keep up with the mustard.

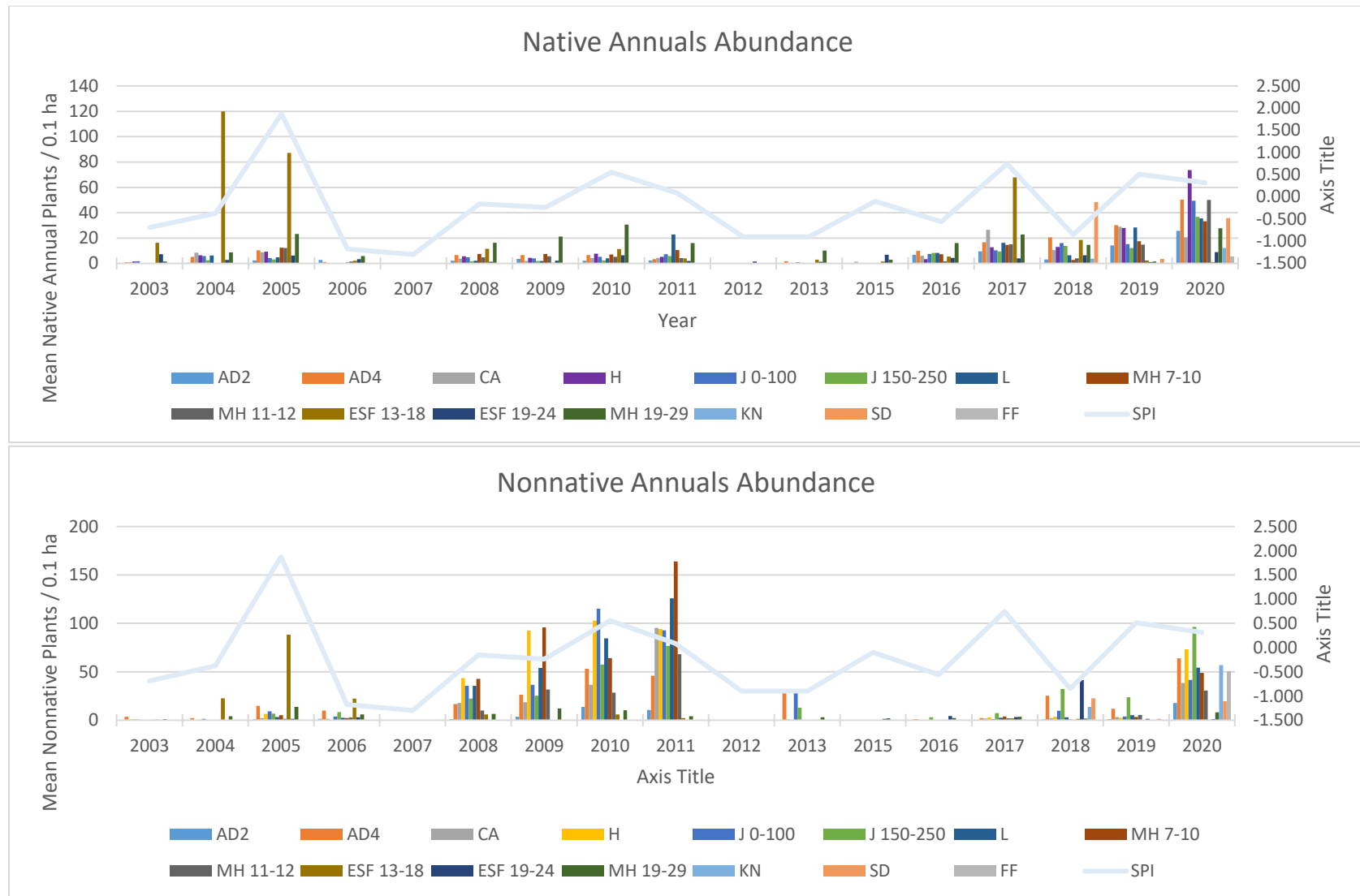


Figure 11: Mean native and nonnative plant abundance since 2003 by plot cluster. KN = Kim Nicol Trail, SD = Stebbins Dune, FF = Fingal's Finger. We began sampling at L in 2004, MH 7-10 and MH 11-12 in 2005, KN and SD in 2018, and FF in 2020. We did not survey KN in 2019. ESF 7-12 is not shown because we did not sample here in 2019 or 2020. The blue line indicates winter SPI.

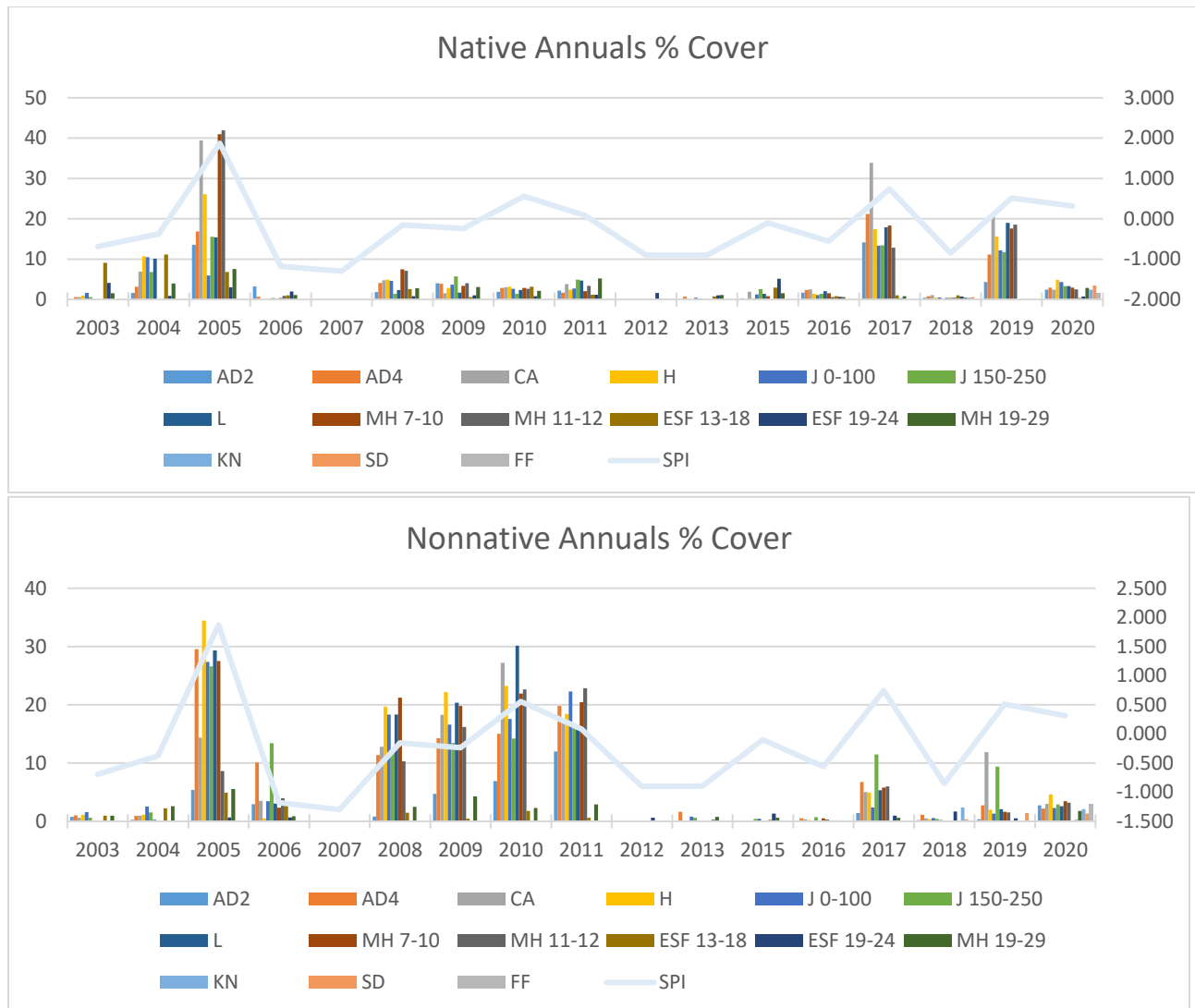


Figure 12 Mean native and nonnative plant cover since 2003 by plot cluster. KN = Kim Nicol Trail, SD = Stebbins Dune, FF = Fingal's Finger. We began sampling at L in 2004, MH 7-10 and MH 11-12 in 2005, KN and SD in 2018, and FF in 2020. We did not survey KN in 2019. ESF 7-12 is not shown because we did not sample here in 2019 or 2020. The blue line indicates winter SPI.

From 2008 to 2011, there was significantly higher invasive annuals coverage on the mesquite dune, active dune and stabilized sand field sites (Figure 12). Those same community types increased their species richness in 2016 through 2019, when there was less coverage of non-natives (Figure 5). Also notable is the steady increase in species richness at our westernmost survey site, ESF19-24, since 2012. The cause of this increase in diversity was not associated with reductions in non-native species and is unknown, but may be due to changes in precipitation patterns, temperatures, and/or changes in levels of sand activity.

Arthropod Monitoring

Introduction

Pitfall traps have been widely used by entomologists since the late 1800s. Pitfall traps are a passive (does not require constant monitoring) trap design used to sample ground-dwelling arthropods. Extensive variation exists in pitfall trap design, but the basic blueprint involves setting a collecting container, which can be anything from a small glass vial to a large bucket, into the ground so that wandering arthropods will unintentionally fall into the collecting container where the researcher can later collect them. Pitfall traps are invaluable research tools because they require minimal effort to install and maintain and they allow for the study of an otherwise largely undetectable yet extremely important animal assemblage.

Our primary objectives are to 1.) document how species' abundance changes over time and correlate this to changing landscapes, such as loss of sand and increase in invasive plant cover, 2.) identify arthropod species or species assemblages that we can use to help characterize habitat types, and 3.) monitor changes in harvester ant abundance, a critical food source for flat-tailed horned lizards (*Phrynosoma mcallii*) and Coachella Valley fringe-toed lizards (*Uma inornata*).

Methods

Every spring, we measure ground-dwelling arthropod species richness and abundance across our aeolian community plots using non-lethal pitfall traps. Each trap consists of a single plastic 1-liter deli cup, funnel, and shade cover (Figure 13). We sink the deli cup into the ground so that the top of the cup is flush with ground level, and then we place a funnel into the top of the cup, preventing escape of captured arthropods. We use a small Masonite board elevated above the trap by wooden pegs to provide shade for captured insects and camouflage from animals that might tamper with the traps, such as ravens. Wandering arthropods encounter the trap and fall into the cup where they remain until we arrive the next day to collect the pitfalls. To record the contents of the traps, we remove the cups from the ground and dump the contents onto a light-colored surface such as a pillowcase or white fiberboard. We then record the sampled species and abundance with the assistance of magnifying loupes and aspirators. We release captured nocturnal arthropods into a shady spot so they are not injured by the temperatures experienced in direct sun.

Each plot hosts 3 pitfall traps – one trap per 0m, 50m, and 100m mark. We set traps for an approximately 24-hour period and intentionally select sampling periods which have low wind to minimize the risk of traps being filled by blowing sand. To further minimize the risk of blowing sand, we often place traps on the downwind side of shrubs which provide some protection. We conducted trapping from April 23 to June 16. There were few windless nights during the sampling period, so we were unable to set traps at AD2, MH 7-12, and Fingal's Finger.

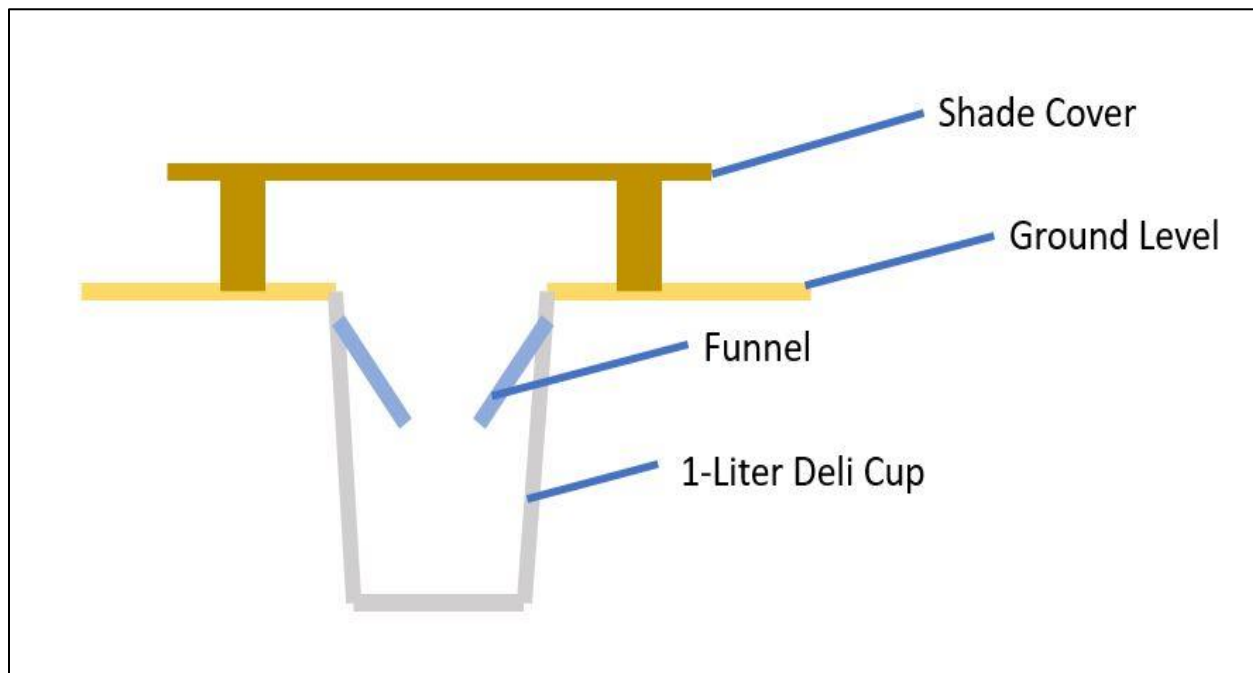


Figure 13: pitfall trap components.

Arthropod Diversity

Insect diversity on sand dunes can be surprisingly high. To date, we have sampled dozens of species across 14 insect orders. Beetles (order Coleoptera) are the most species-rich group sampled with our pitfalls, with most of the richness owing to darkling beetle (family Tenebrionidae) diversity. Ants, bees, and wasps (order Hymenoptera) are another important group sampled with our pitfalls. Minute parasitic wasps (e.g. superfamilies Chalcidoidea and Platygastroidea) show a high diversity but are relatively rarely sampled. Ants (family Formicidae), on the other hand, show lower species richness but an extremely high abundance, and are therefore treated as one of the most ecologically important insects present. As such, we

have selected ants and darkling beetles to serve as potential indicator species for habitat type and health across our aeolian plots.

Ants

Commonly sampled ants include California harvester ants (*Pogonomyrmex californicus*), big-eye harvester ants (*Pogonomyrmex magnacanthus*), black harvester ants (*Veromessor pergandei*), Kennedy's honeypot ant (*Myrmecocystus kennedyi*), *Myrmecocystus tenuinodis*, *Dorymyrmex* sp., and *Forelius pruinosus*. Harvester ants are often considered keystone species due to their capacity to harvest large quantities of seeds, which serve as their primary food source, therefore affecting plant abundance and distribution. Ant nest excavations also provide aeration and nutrient transport for the soil they inhabit. Importantly, harvester ants are also a primary or important food source for flat-tailed horned lizards and Coachella Valley fringe-toed lizards. It was found in our 2019 assessment of dune ant communities that different aeolian habitat types can be distinguished by the species of ants present. This is likely due in part to certain ants' tolerance of particular substrate types (e.g. *Veromessor* and *Dorymyrmex* preferring coarse sand/gravel).



Figure 14 shows the mean number of harvester ants (*P. californicus*, *P. magnacanthus*, and *V. pergandei*), captured per pitfall by year since 2008. *P. californicus* are extremely abundant on the Coachella Valley National Wildlife Refuge (CVNWR), represented here by the stabilized sand field and active dune sites. Conversely, *V. pergandei* is extremely rare at the

CVNWR. The opposite is true for the ephemeral sand field sites, located at the Whitewater Floodplain Preserve – *V. pergandei* is the dominant harvester ant, with moderate amounts of *P. californicus* and low numbers of *P. magnacanthus*. Since 2008, *P. californicus* has displayed signs of a general downward-trend on the CVNWR. This ant exists across a wide range of substrate types and therefore it is not expected to be severely affected by sand stabilization. However, because this species depends primarily on seeds as a food source, a change in either the abundance or quality of available seeds may be able to noticeably affect population size or alter activity.

Sahara mustard (*Brassica tournefortii*) has established on the CVNWR as a significant invasive threat. If sufficient early winter rains are present, this plant has the ability to quickly establish and outcompete native annuals. A large mustard plant can also produce thousands of seeds, potentially saturating the seed bank and altering the diversity of seeds available for ants to harvest. We have noted that *P. californicus* will harvest mustard seeds, but it is unknown if these seeds are actually palatable or if they provide the same nutrition as other native seeds. Alternatively, it is also possible that the mustard seeds provide an adequate food source for the ants, and due to the large amount of mustard seed produced, it may be easier for the ants to find food, resulting in less time needed to spend foraging and therefore less encounters with our pitfalls. The latter scenario would superficially appear to indicate that the ants are disappearing, but instead they are actually thriving and only their activity pattern has changed. However, we are in the process of conducting ongoing laboratory experiments investigating the effects of diet on *P. californicus* colony founding, and preliminary results suggest that queens do not recognize mustard seed as a viable food source. Even if established colonies are able to utilize mustard seed as a food source, newly emerged queens may not be able to use this seed and are therefore having a harder time establishing new colonies. Also, it is worth noting that 2020 is the first year since 2012 that *P. magnacanthus* were not sampled at the CVNWR. However, this ant usually occurs at relatively low numbers, so the absence may be due to chance.

We used PC-ORD (Wild Blueberry Media LLC) to investigate the potential for using ant species assemblages as a means to differentiate aeolian habitat types. We compiled data for mean number of ants per trap for all years since 2008 and used PC-ORD to analyze the similarity of these habitat types using the seven most abundant ant species. Figure 15 shows the DCA ordination results for all mean abundance data available since 2008 for *P. californicus*, *P.*

magnacanthus, *V. pergandei*, *M. kennedyi*, *M. tenuinodis*, *F. pruinosis*, and *Dorymyrmex* sp. Active dune and stabilized sand field sites (CVNWR) cluster tightly against Axis 2 due to the dominance of *P. californicus* at these sites, as demonstrated in Figure 15. Not far from this cluster are the Kim Nicol, Fingal's Finger, Stebbin's Dune, and Mesquite Hummock sites. Again, the similarity here is driven largely by an abundance of *P. californicus*. However, the ephemeral sand field sites stand fairly distinctly apart, with the Tipton Road plot demonstrating a higher abundance of *M. kennedyi* and *Dorymyrmex* sp., and the Whitewater Floodplain Preserve plots showing greater numbers of *V. pergandei* and *M. tenuinodis*. While both these sites are considered to be ephemeral sand fields, the Whitewater Floodplain tends to have coarser sand, more gravel, and less vegetation cover. These differences likely result in the observed separation of ant assemblages.

Figure 16 shows the DCA ordination results for the same ant species separated into four temporal groups: 2008-2010, 2011-2013, 2015-2017, and 2018-2020. The purpose of this analysis is to visualize any shifts in species assemblages that may have occurred since 2008. The results are mostly similar to those shown in Figure 15, indicating that these sites have remained fairly stable in their species composition. One exception are the mesquite hummock plots at Willow Hole, which do not appear to strongly cluster, and to a lesser extent the same is true at the Whitewater Floodplain plots. While the exact cause of this variability is unknown, these results may indicate that these aeolian communities may experience more variable conditions (wind, amount of sand, annual cover, etc.) than areas such as the CVNWR, which periodically affects ant abundance and/or activity.

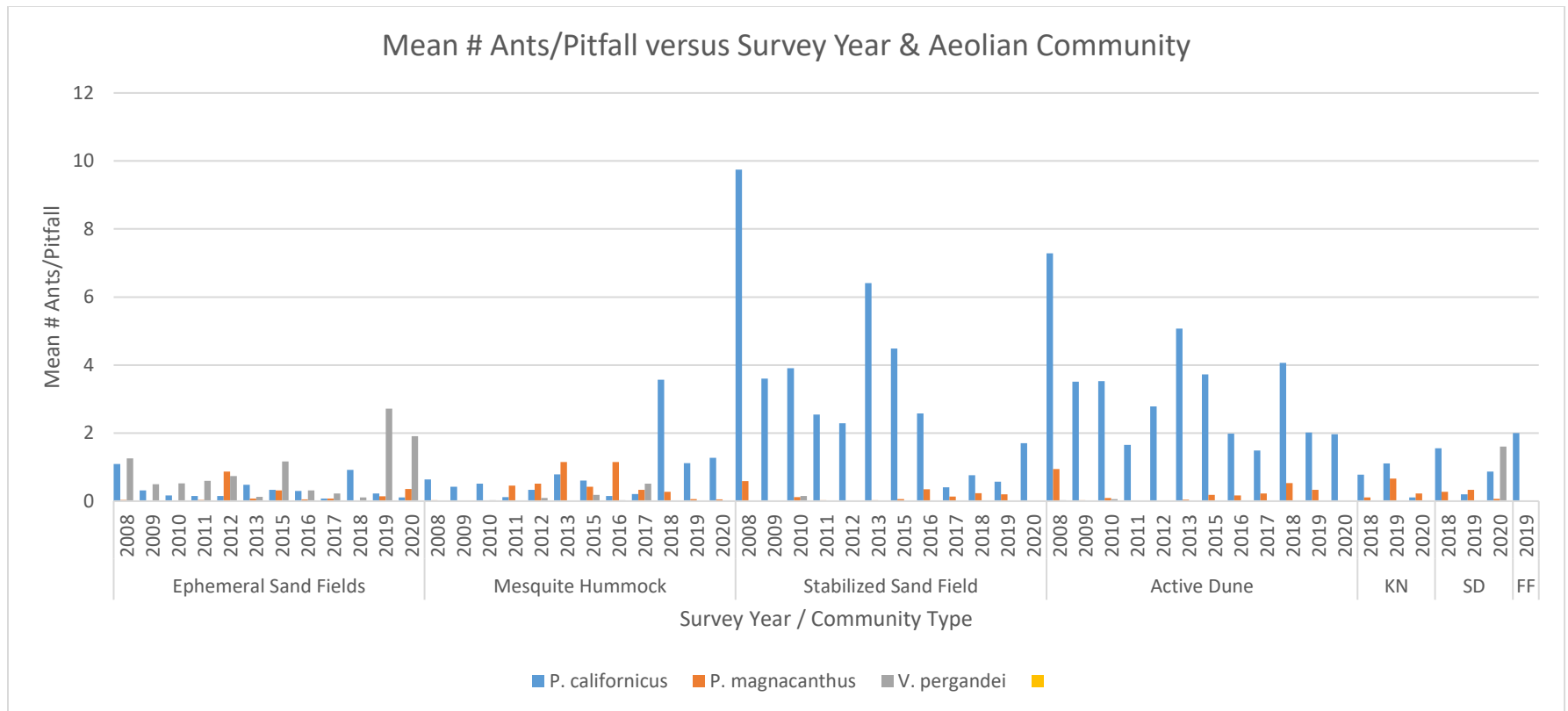


Figure 14: pitfall survey results for harvester ants (*Pogonomyrmex californicus*, *P. magnacanthus*, and *Veromessor pergandei*) since 2008, by aeolian community type. KN = Kim Nicol Trail, SD = Stebbin's Dune, FF = Fingal's Finger.

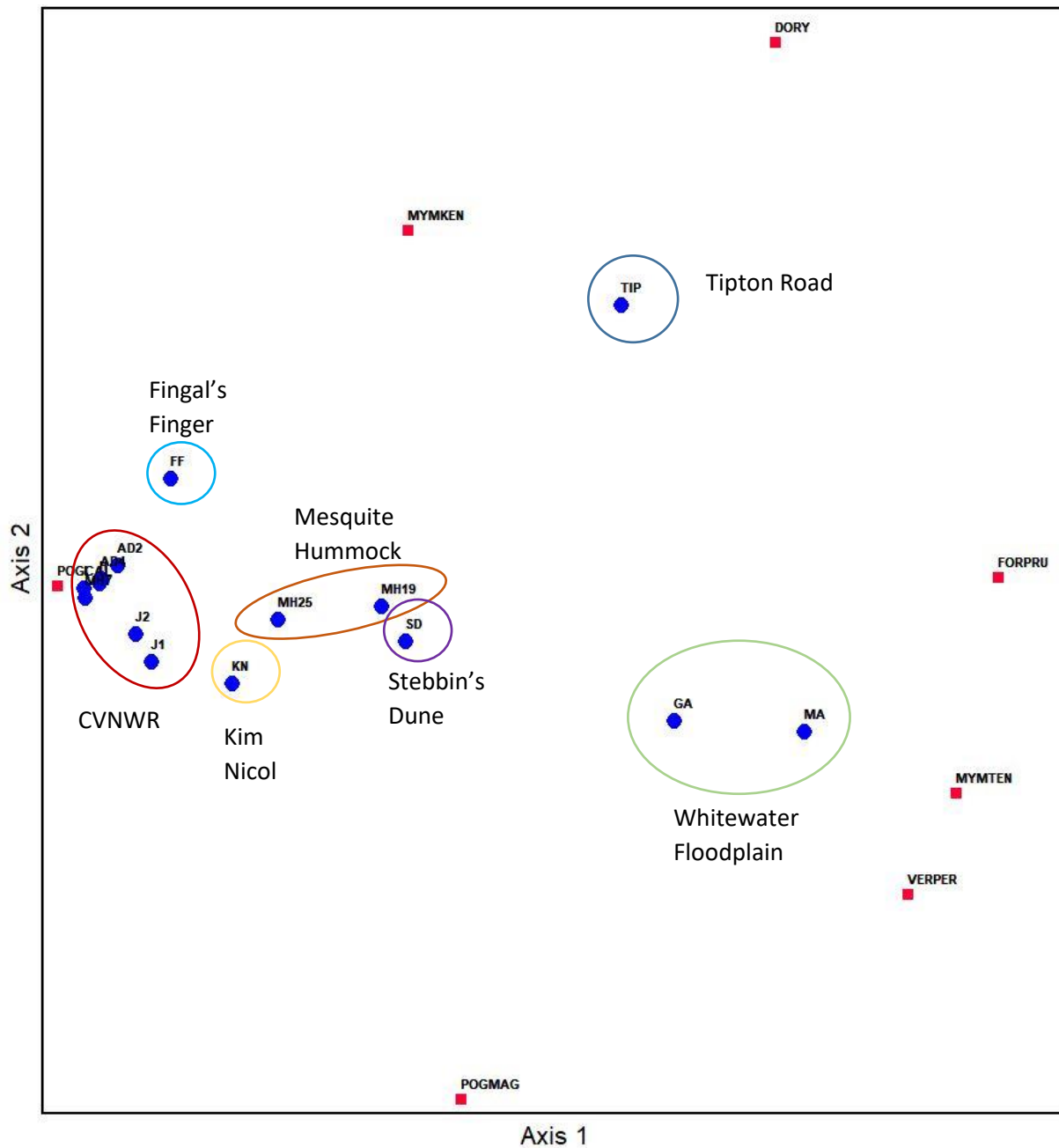


Figure 15: DCA ordination analysis showing separation between plot clusters based on mean ants per pitfall data for seven ant species since 2008. Distinct localities are circled. POGCAL = *Pogonomyrmex californicus*, POGMAG = *P. magnacanthus*, MYMKEN = *Myrmecocystus kennedyi*, MYMTEN = *M. tenuinodis*, DORY = *Dorymyrmex* sp., FORPRU = *Forelius pruinosus*, VERPER = *Veromessor pergandei*.

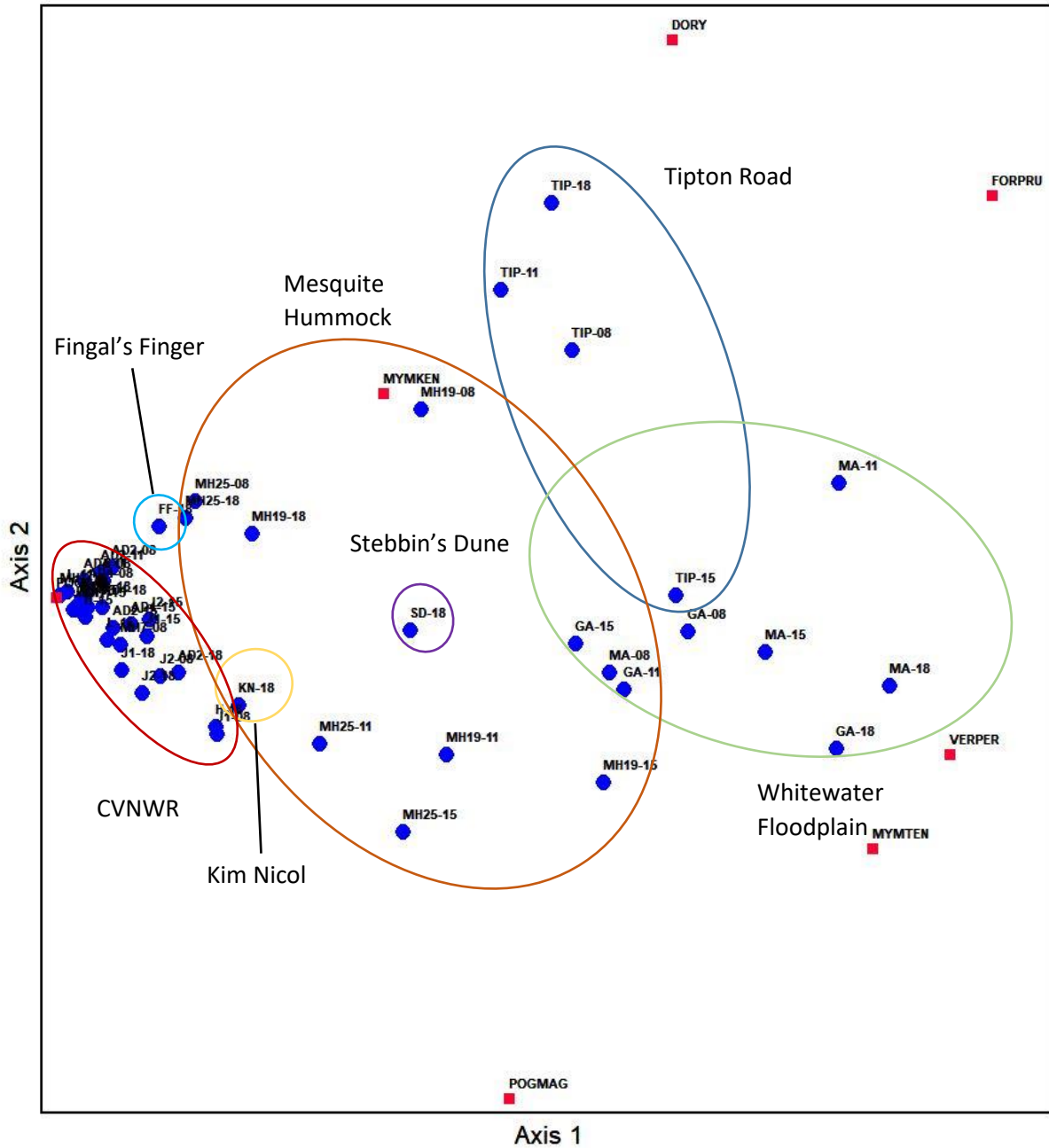


Figure 16: DCA ordination analysis showing separation between plot clusters based on mean ants per pitfall data for seven ant species since 2008, separated into 3-year groups. Plot point titles with suffix -08 = 2008-2010, -11 = 2011-2013, -15 = 2015-2017, -18 = 2018-2020. Distinct localities are circled. POGCAL = *Pogonomyrmex californicus* (left-most red point, label obscured), POGMAG = *P. magnacanthus*, MYMKEN = *Myrmecocystus kennedyi*, MYMTEN = *M. tenuinodis*, DORY = *Dorymyrmex* sp., FORPRU = *Forelius pruinosus*, VERPER = *Veromessor pergandei*.

Darkling Beetles



Darkling beetles are diverse both in their species richness and their morphology. Some species, such as the armored stink beetle (*Eleodes armata*), are among the largest insects present at up to 3.5 centimeters in length, while species such as *Batulius setosus* are only about 4 millimeters in length. Most species are nocturnal foragers, but others, such as *Araeoschizus hardyi*, are found in association with ants and likely feed off of their food stores. Two of the larger species, the blue death-feigning beetle (*Asbolus verrucosus*) and the smooth death-feigning beetle (*Asbolus laevis*) are very common in their respective habitats. *A. verrucosus* are found in more stabilized areas, such as creosote scrub surrounding dunes and basins dominated by herbaceous plants. *A. laevis* appear to prefer exclusively habitat characterized by loose, deep, active sand, such as active dunes and mesquite dunes. This difference in substrate preference makes them potentially useful for identifying changes in sand compaction and stabilization, with less *A. laevis* being detected at a site with increasing stabilization.

Figure 17 shows the mean number of death-feigning beetles sampled per pitfall per year. As predicted, sites characterized by fine, loose sand, such as active dunes, have an extremely high abundance of *A. laevis*, which appears better adapted for these habitats. *A. verrucosus* is found in every habitat type, although it is typically not found regularly on large dunes, but rather around the margins. We have never sampled *A. laevis* from an ephemeral sand field site (Whitewater Floodplain Preserve or Tipton Road plots), and we have only sampled six *A. verrucosus* here since 2008 (not shown on graph). Large beetles appear to be relatively rare at ephemeral sand field sites, however *Eleodes armata* is slightly more common here than *Asbolus* species. Similar to *P. californicus*, *A. laevis* appears to be displaying a downward-trend on the

CVNWR active dune sites, as well as at the Kim Nicol Trail, which is also largely active dunes. Because *A. laevis* is strictly tied to loose sands, this decline may indicate gradual stabilization of the dunes, however there are many other variables that have not been investigated, such as food sources, predation, water availability, etc., that may also explain this trend.

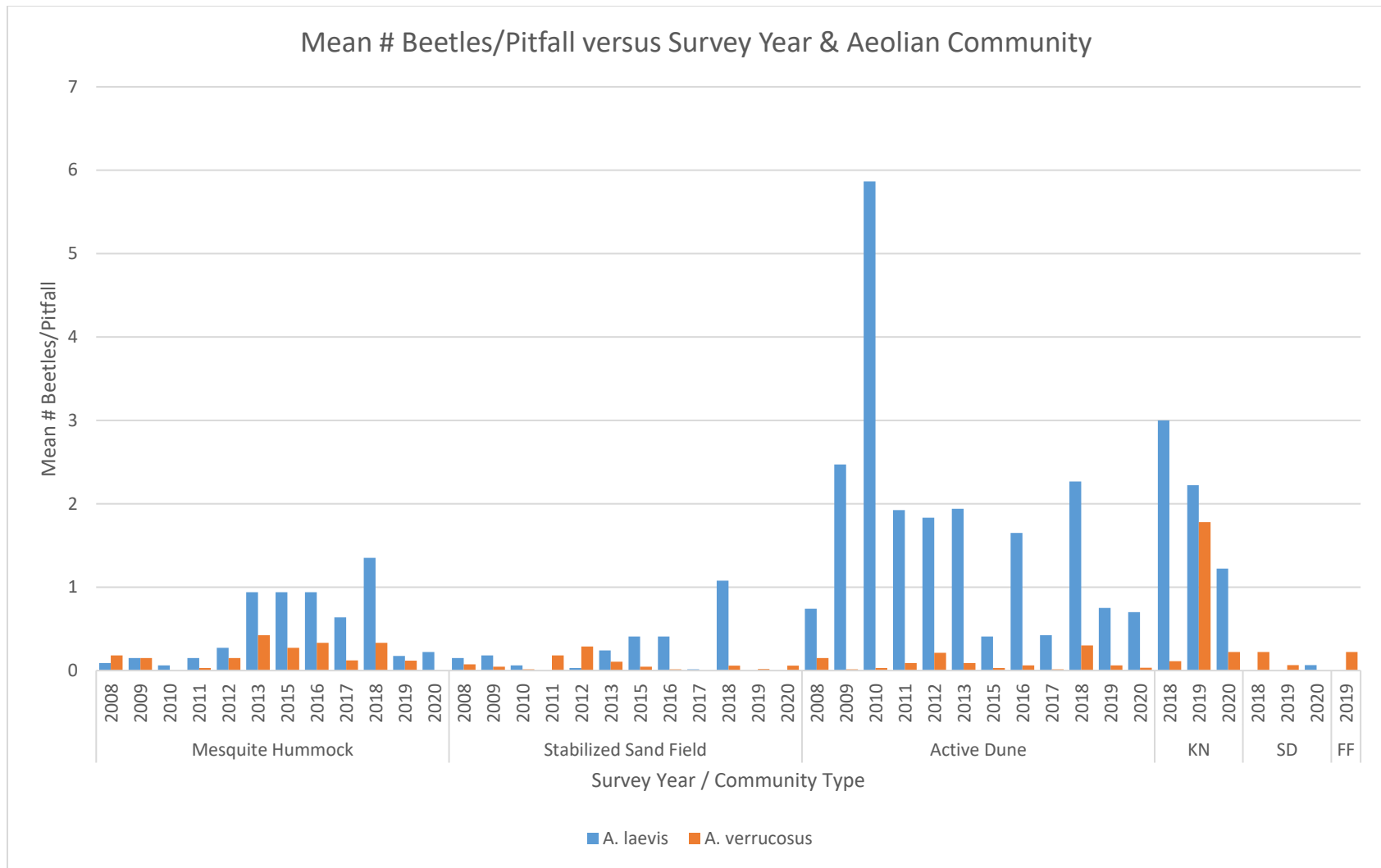


Figure 17: pitfall survey results for death-feigning beetles (*Asbolus laevis* and *A. verrucosus*) since 2008, by aeolian community type. KN = Kim Nicol Trail, SD = Stebbin's Dune, FF = Fingal's Finger.

Round-tailed Ground Squirrel

Round-tailed ground squirrels (RTGS), *Xerospermophilus tereticaudus chlorus*, occur in fine-textured sandy areas of the Coachella Valley. Antelope ground squirrels replace RTGS in gravelly and rocky soils. RTGS are mostly restricted to aeolian sands, and occur throughout the valley's aeolian sand communities, as well as in urban gardens along wildland-urban interfaces where soils are appropriate. Our survey method, similar with all the vertebrates included here, is to quantify their abundance based on the mean number to their distinctive track ways left within our 0.1 ha plots. Unlike other (non-avian) vertebrates, RTGS are quite vocal when occurring at high densities; there we use their distinctive alarm calls and tracks (whichever provides the higher number) to tabulate occurrences within our plots. However, at low densities, they rarely vocalize, and so we can only use their tracks for surveys. Within the protected aeolian sand habitats of the CVMSHCP RTGS are fairly rare, except within the mesquite dunes of the Willow Hole Preserve, where they can be abundant.

Within the mesquite dunes, they show year-to-year variation in numbers that roughly correlate with annual precipitation (Figure 18). The explanation for the lack of a stronger rainfall response is that the mesquite are typically tapped into aquifer-based water sources and not reliant on annual rainfall. In areas where the mesquite have died, RTGS densities drop to match those on non-mesquite aeolian communities.

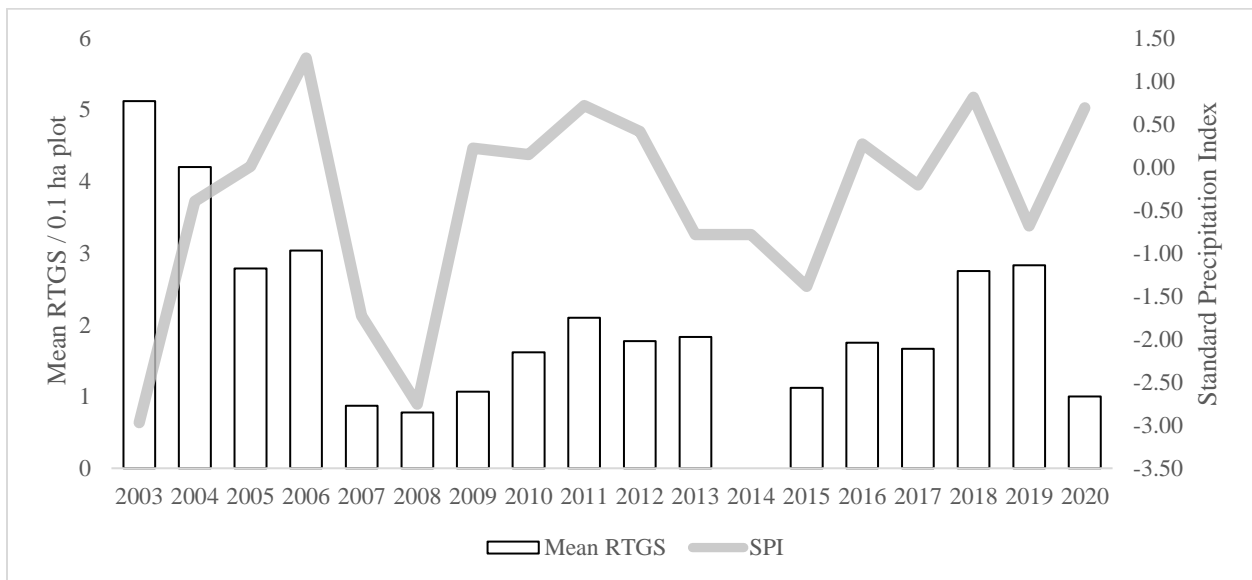


Figure 18. Temporal patterns of abundance of round-tailed ground squirrels within the Willow Hole Preserve. The SPI is off-set by one year to account for the one year lag time most vertebrate show between rain and population responses. We did not collect data in 2014 at the request of the wildlife agencies.

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Appendix VI –
2020 Monitoring Results for Burrowing
Owls and Palm Springs Pocket Mice



Coachella Valley Conservation Commission

2020

Coachella Valley Multiple Species Habitat Conservation Plan

& Natural Community Conservation Plan

2020 Monitoring Results for Burrowing Owls and Palm Springs Pocket Mice



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Burrowing Owls

INTRODUCTION

Burrowing owls (*Athene cunicularia*, hereafter BUOW) are wide-ranging raptors found throughout much of the Western Hemisphere (Rosenberg & Haley 2004). BUOWs are found throughout the deserts of California, however, most records are associated with agriculture or other irrigated landscapes. The association with irrigated landscapes may be real or may in part reflect a bias as to where people spend time looking for the owls. In Southern California's Coachella Valley, BUOW are one of the 27 protected species under the Coachella Valley Multiple Species Habitat Conservation Plan (CVMSHCP). In the neighboring Imperial Valley, historical owl records were sparse. More recently, in parallel with the development of agricultural operations and supporting irrigation networks, BUOW population densities are now among the densest that have been documented (DeSante et al. 2004, Trulio & Higgins 2012). In the Coachella Valley, BUOW often occur in natural desert habitats, usually in association with deep-cut washes and surrounding upland areas. These washes also usually support larger populations of burrowing mammals, most importantly ground squirrels (*Xerospermophilus tereticaudus* and *Ostospermophilus beecheyi*), that are essential for creating burrows that the owls can later occupy (Klute et al. 2003, Gervais et al. 2008, Conway 2018). Similar to the Imperial Valley, BUOW are also common along stormwater/wastewater channels within agricultural and urban landscapes. Within this range of habitats BUOW prey upon a broad variety of vertebrates (mostly small mammals) and arthropods within the American Southwest (Barrows 1989, York et al. 2002, Hall et al. 2009, Ayma et al. 2019).

Despite the burrowing owl's apparent flexibility in occupying habitats with varying degrees of human influence, studies have demonstrated adverse effects related to human activities. Reduction of open spaces, including agricultural areas, through urbanization (Millsap & Bear 2000, Gervaise et al. 2008, Berardelli et al 2010), altering of available prey species (York et al 2002, Trulio & Higgins 2012), increased risk of mortality by predation (White 1998, Conway et al. 2006) and vehicle collisions (Klute et al. 2003, Berardelli et al. 2010), and reduction in burrowing mammals that contribute to nest-starting (Coulombe 1971, Desmond et al. 2000) are among the factors influencing the stability of urban owl populations.

Nevertheless, BUOWs occupying natural and agricultural habitats in both the Coachella Valley appear to be sustaining populations there. Here we took advantage of the gradient of human influences across the habitats that BUOWs occupy in the Coachella Valley to describe metrics related to their apparent sustainability along that gradient. In the spring of 2015, we began monitoring local burrowing owl populations along a gradient of habitat disturbance throughout the Coachella Valley. Except that no surveys occurred in 2016, each year since 2015, through 2020, we assessed three main metrics that provided insights into the relative sustainability of the Coachella Valley's BUOW populations. Those metrics included: 1) reproductive success, 2) nest disturbances such as predation or human interference, and 3) variability in prey selection/availability.

METHODS

The three locations we chose for this study were in the Coachella Valley Preserve (on conserved Bureau of Land Management land in Sky Valley with very little human disturbance, hereafter Sky Valley or SV), Little Morongo Wash in Desert Hot Springs (moderate human disturbance but fairly intact natural habitat, hereafter Desert Hot Springs or DHS), and several locations in the East Valley near Coachella (very little natural habitat, replaced with mostly agricultural and urban development, hereafter East Valley or EV).

Desert Hot Springs

The burrows we monitored in Desert Hot Springs (DHS) in 2020 were within the Little Morongo Wash between 15th Avenue to the south and Ironwood Drive to the north (Figure 1). All the burrows we monitored this year were also monitored at least once in a previous year. Compared to other owl-occupied areas in the Coachella Valley, such as the stormwater channels and agricultural areas in the East Valley, the Little Morongo Wash site represented relatively natural habitat. However, this site was surrounded by a patchwork of urban areas, both residential and commercial, separated by areas of natural or moderately disturbed habitat, and thus human disturbances are common. We frequently observed illegal dumping, illegal off-road recreation, homeless encampments, and human foot traffic.

The Morongo Wash was over 100 meters wide in some spots and contains numerous areas where floodwater had cut vertical wash sides roughly 1.5 meters in height. These wash sides were frequently occupied by burrowing owls and nest-starting round-tailed ground squirrels. Wash-bottom vegetation included black-banded rabbitbrush (*Ericameria paniculata*), cheesebush (*Ambrosia salsola*), and sparse desert willow (*Chilopsis linearis*). The upland landscape surrounding the wash is dominated by creosote bush (*Larrea tridentata*) and burrobush (*Ambrosia dumosa*). The federally endangered Coachella Valley Milkvetch (*Astragalus lentiginosus* var. *coachellae*) was also commonly found in the upland areas near the burrow sites.

Sky Valley

In 2019 we learned of burrowing owl nests previously identified on Bureau of Land Management land in a broad wash north of Thousand Palms Oasis and south of Dillon Road. We relocated nests in this area and monitored three burrows. We decided to monitor this area again in 2020 because it represents, to our knowledge, the least disturbed and most accessible area with a high density of owl burrows. Two of the burrows we monitored this year were also studied in 2019. We also added two new burrows, located on the opposite side of the wash from the original burrows, which we discovered this year.

Most of the burrows we monitored in this area are located along the margins of a broad wash. These burrows were either constructed into a steep slope or on a nearly vertical cliff face. Dominant vegetation in the wash was Schott's indigo (*Psoralea schottii*), sandpaper plant (*Petalonyx thurberi*), and cheesebush. One of the burrows was in nearby upland habitat

dominated by creosote bush and burrobush. The wash and, especially, the upland habitat possessed a much higher cover of stone/cobble and gravel than the DHS site.

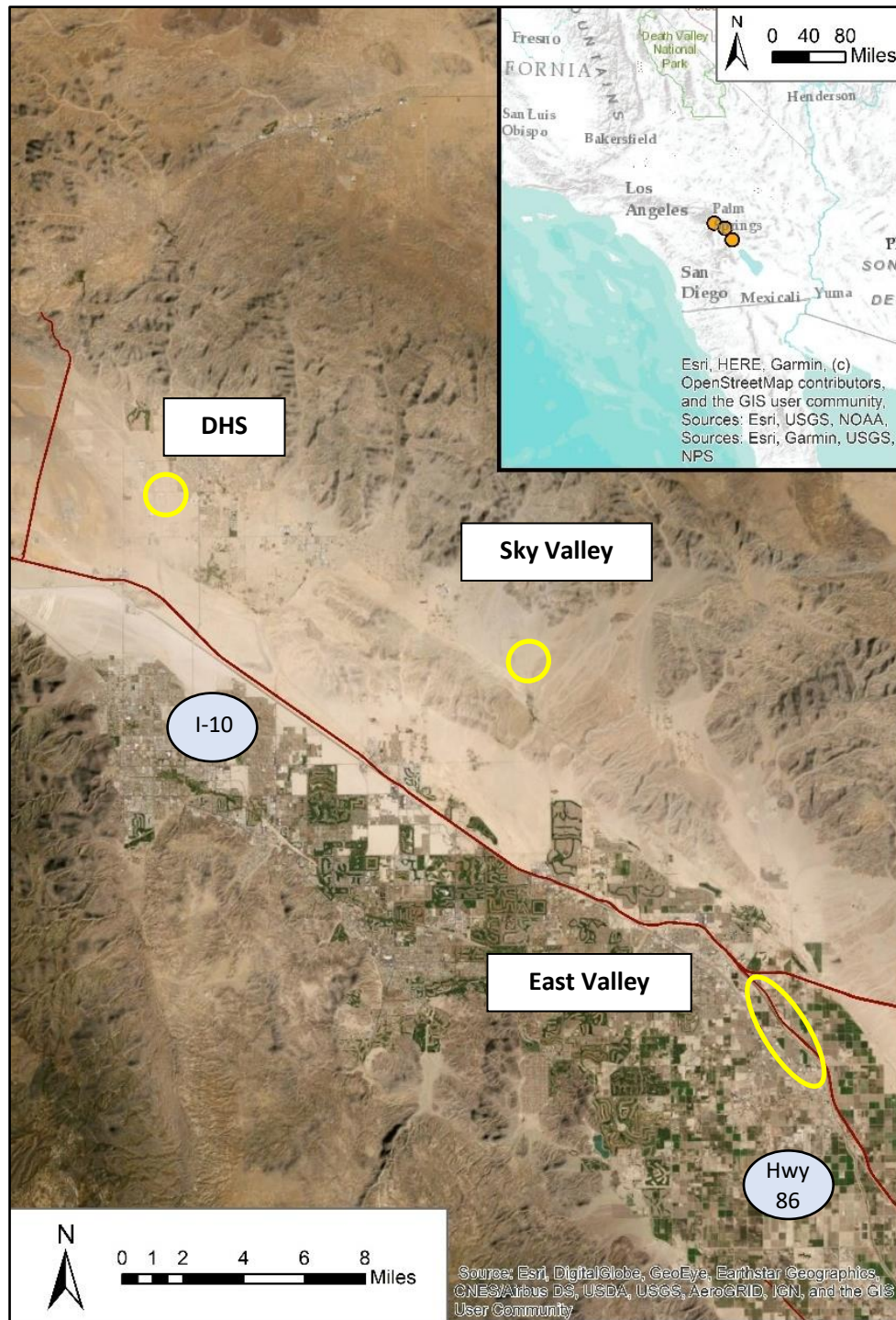


Figure 1: map of Coachella Valley showing broad locations of our three study sites.

East Valley

Three nests that we monitored in 2018 and 2019 were located on land owned by the Twentynine Palms Band of Mission Indians. We jointly monitored these burrows along with the San Diego Zoo Institute for Conservation Research. One of these burrows (EV-2) was artificial. Our study sites in the East Valley were generally characterized by a high level of human disturbance in the form of agriculture and urbanization. All nests we monitored here were either located on or in very close proximity to a large wastewater channel that typically contain a source of perennial water. Additionally, either the I-10 freeway or Highway 86 were adjacent to all monitored burrows. The substrate in the East Valley was typically composed of extremely fine and saline ancient lakebottom clay and silt. Native vegetation was scarce here but was occasionally found in parallel to the wastewater channels in the form of dense honey mesquite (*Prosopis glandulosa* var. *torreyana*) thickets and basins containing salt-tolerant species such as iodine bush (*Allenrolfea occidentalis*) and saltbushes (*Atriplex* spp.). The bottoms of the wastewater channels exhibited riparian flora, most prominently cattails (*Typha* sp.), but also occasional cottonwood trees (*Populus deltoides*). Invasive tamarisk (*Tamarix* sp.) was found amongst natural vegetation and artificial riparian habitats. The riparian areas and bordering channel slopes were regularly subject to massive disturbance in the form of maintenance grading and clearing of vegetation. We did not monitor this area in 2020 due to the theft of several cameras in 2018 and 2019.

Data Collection

Our data collection in 2020 involved two main components: continuous monitoring of nest activity via camera “trap” (Bushnell NatureView Cam HD Model 119740), and collection of regurgitated pellets for studying owl diets. These heat and motion triggered wildlife cameras were also able to film at night using infrared. We set the cameras to capture three photos (one every second) once triggered, and the cameras were set to be triggered no more than once every 30 seconds. We always placed cameras about one to two meters in front of the burrow entrance to avoid providing an unseen perch for predatory birds. In DHS, due to increased risk of theft or tampering, we placed cameras inside locked metal lockboxes which we attached with screws to the tops of metal t-posts. At our Sky Valley monitoring sites, there was little risk of human tampering, so we placed the cameras inside unlocked lockboxes which we attached to short wooden stakes. We began camera monitoring during the BUOW breeding season, from April 16 to April 27 to June 16 to June 23 (see Table 3 for exact dates). We visited cameras once per week to collect pellets, photos, change batteries, and make any necessary adjustments to the camera settings.

We collected pellets from all burrows during each camera maintenance visit. We dissected the pellets in our lab and identified prey items to the lowest taxonomic level possible. We used bones (mostly mandibles) to identify vertebrates, and pieces of exoskeleton, usually either heads or hard mouthparts (mandibles or parts of chelicerae) that survived digestion, to identify invertebrates. Because some pellets can remain intact for weeks or months, on our first

visit to install cameras we cleared away all existing pellets to avoid an inaccurate temporal representation of prey items.

Invertebrate prey remains were largely identified through comparison with preserved specimens. Occasionally, this was supplemented with online resources such as BugGuide (<https://bugguide.net>, Iowa State University Department of Entomology). Vertebrate remains were identified using Hall (1981) and by comparison to preserved specimens.

Prey item live weights were estimated using the following methods: 1.) measurements from captured animals (several tenebrionid beetle species); 2.) peer-reviewed publications for scarab beetles (Lease et al. 2012), ants (Lighton and Quinlan 1994, Quinlan and Lighton 1999, Tschinkel 2013), grasshoppers (Chappell 1983), Palm Springs Pocket Mice (Bartholomew and Cade 1957), lizards (Dawson and Bartholomew 1956), and shovel-nosed snakes (Sharpe et al. 2015); 3.) reputable online resources, i.e. Animal Diversity Web online database (<https://animaldiversity.org>, University of Michigan Museum of Zoology) and All About Birds (<https://allaboutbirds.org>, Cornell University), and; 4.) estimation of weight based on comparison to a prey species with a known weight and similar size. We decided to analyze the percent of prey species by mass in diet instead of our previous method using the amount of prey items per pellet as our main dietary metric because we believe, even if some prey masses are rough estimates, this method provides a more accurate representation of the total amount each prey species contributes to the owls' diet.

“Reproductive success” is defined in our report as the maximum number of owlets produced per breeding pair. Typically, the number of owlets that survive to fledging is used as a measure of reproductive success, however fledge dates are difficult to accurately identify based on photos alone. Also, we found limited evidence of owlet mortality amongst our study nests, and therefore we believe that, for our study nests, the maximum number of owlets produced and the maximum number of owlets fledged were usually equivalent. As demonstrated by Martin (1973), some burrowing owl populations can exhibit high rates of owlet survival to fledging (in this case, 94.9% of 78 owlets survived to fledging).

We compiled monthly precipitation measurements from two sites: the Palm Springs International Airport (approx. 33°49'N, 116°30'W), hereafter used as a proxy for rainfall at the Whitewater Floodplain Preserve (WWFP), and a HOBO weather station at the Coachella Valley National Wildlife Refuge (CVNWR) (approx. 33°47'N, 116°20'W). When analyzing the effects of annual precipitation versus reproductive success, we used the data collected from the weather station nearest to each burrow (WWFP measurements for DHS burrows and CVNWR measurements for Sky Valley and East Valley burrows).

Statistical Analyses

We only used data collected from breeding nests that yielded a reliable maximum owlet count (no significant camera malfunctions), unless otherwise noted. However, to evaluate the BUOW's diet as a function of their breeding success, we incorporated all burrows, regardless of whether they were breeding or not if the pellet data was deemed sufficient to capture the breadth of the BUOWs diet. We did not include data from 2015 and 2017 in our diet analyses because the level of prey identifications during these years was not comparable to the following years.

Also, we did not individually analyze mean reproductive success data from 2015 (both DHS and East Valley) and 2019 (Sky Valley) due to lacking a sufficient sample nests. However, we included reproductive success data from these sampling efforts when analyzing all nests from a site combined across all monitoring years.

To explore the variation between the diets of BUOW from different sites, we performed ordination analyses and produced ordination graphs and subsequent correlation values using PCord v.6.08 (McCune and Mefford 2011). We used the detrended correspondence analysis (DCA) ordination method with 26 segments. To focus only on the effects of prey items that comprise an ecologically relevant proportion of the owls' diets, we excluded singletons, doubletons, and species that never achieved greater than 1% of total prey mass at any nest, resulting in 21 prey taxa used for the analysis. We selected axes to graph based on which returned the clearest separation of sites and we reported the corresponding correlation coefficients for these axes. We used a plexus overlay to indicate strong and weak associations between prey species. PCord uses a signed standardized chi-square distance measure when evaluating associations. We designated a chi-square score between 0.3 and 0.5 as a "weak" association, and greater than 0.5 as a "strong" association, applying Yates' correction to all frequency values.

To determine correlation between reproductive success and diet and annual precipitation, we used Microsoft Excel to perform Pearson's correlation analyses and subsequent significance tests. To investigate the differences between mean reproductive success and location/time, we used hand-calculated two-tailed Mann-Whitney *U*-tests with significance determined by comparing *U* statistic values to corresponding critical values. *U* statistic values less than or equal to the critical value were deemed significant at a .05 alpha level.

RESULTS

Ordination

Our DCA ordination analyses revealed distinct clustering by nest site based on the species and abundance of various prey items in the owls' diet. Figure 2a shows the results of our analysis using the percent of total prey mass per prey species in the diet. This analysis produced three fairly distinct groupings based on location: DHS, Sky Valley, and East Valley storm channels. The most significant correlations to Axis 1 are *Edrotes* beetles (Tenebrionidae), scarab beetles (Scarabaeidae), cerambycid (longhorn) beetles (Cerambycidae), Palm Springs pocket mice (*Perognathus longimembris* ssp. *bangsi*), desert pocket mice (*Chaetodipus penicillatus*), Merriam's kangaroo rats (*Dipodomys merriami*), and woodrats (*Neotoma* sp.) (Table 1). Association lines between prey species also reveal expected groupings: *Cryptoglossa muricata* beetles (Tenebrionidae), *Eleodes armata* beetles (Tenebrionidae), lizards, Palm Springs pocket mice, woodrats (*Neotoma* sp.), and Merriam's kangaroo rats form an association close to the DHS and Sky Valley sample points, indicating that these species are more closely associated with the diet of owls living in natural habitat, while cerambycid beetles, spiders, earwigs (Dermaptera), harvest mice (*Micromys* sp.), and birds form an association close to the East

Valley storm channel sample points and comprise an important diet component for owls living in more anthropogenic habitats. Western shovel-nosed snakes (*Chionactis occipitalis*) are also important for owls in DHS, and crickets, grasshoppers, and cockroaches (Orthoptera and Blattodea), scarab beetles, *Edrotes* beetles, caterpillar hunter beetles (*Callisoma parvicole*), and Botta's pocket gophers (*Thomomys bottae*) are additional important prey items for East Valley storm channel owls. Wind scorpions (Solifugae), scorpions, cactus mice (*Peromyscus eremicus*), and desert pocket mice are an important dietary component for most owls, regardless of location.

Our comparison analysis, using mean individuals of prey species per pellet instead of percentage of total mass of prey species, is shown in Figure 2b. We included this analysis to compare our previous method used to analyze BUOW diet (using the mean number of prey species per pellet) against our newly implemented method which uses estimated prey mass. Ultimately, the per-pellet method appears to perform similarly to our current method when analyzed using DCA ordination, but lacks the ability to infer the actual importance of each prey species to the owls' nutrition as extremely abundant prey items, such as fire ants, will have a much higher per-pellet abundance than kangaroo rats, but in reality contribute a negligible amount to the total nutritional input due to their minute size.

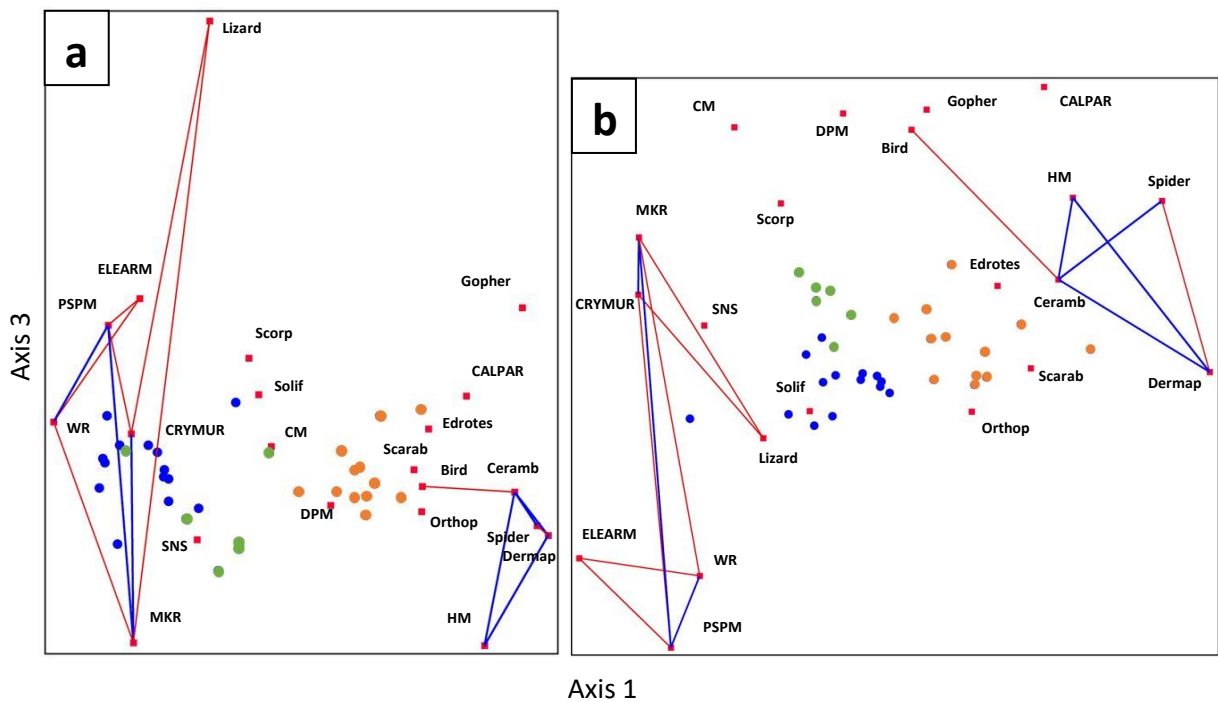


Figure 2: Results of Detrended Correspondence Analysis (DCA) ordination using 21 prey taxa for (a) percent of total prey mass and (b) mean individuals per pellet, excluding singletons, doubletons, and species that never comprised >1% total prey mass. Blue dots indicate DHS nests, green dots indicate Sky Valley nests, and orange dots indicate East Valley nests. Red squares indicate individual prey taxa: ELEARM = *Eleodes armata*, CRYMUR = *Cryptoglossa muricata*, CALPAR = *Calisoma parvicolle*, Ceramb = cerambycid beetles, Dermap = dermapterans), Orthop = orthoperans/blattodea, Scorp = scorpions, Solif = Solifugae), WR =

woodrats, MKR = Merriam's kangaroo rats, PSPM = Palm Springs pocket mouse, DPM = desert pocket mouse, CM = cactus mouse, HM = harvest mouse, Gopher = Botta's pocket gopher, SNS = western shovel-nosed snake. Blue lines indicate strong associations. Red lines indicate weak associations.

Table 1: Correlation coefficients (Pearson's r and Kendall's τ) for ordination axes displayed in Figure 2. Bolded values represent r values greater than .4 or less than -.4, which serves as an indication of strong correlation. See Figure 2 description for explanation of prey species names.

	Abundance Per-Pellet (Figure 2a)						Percent by Mass (Figure 2b)					
	Axis 1			Axis 3			Axis 1			Axis 3		
	r	r2	tau	r	r2	tau	r	r2	tau	r	r2	tau
ELEARM	-.256	.065	-.245	.223	.050	.287	-.476	.227	-.253	-.499	.249	-.378
CRYMUR	-.456	.208	-.413	.021	0	-.088	-.494	.244	-.531	-.083	.007	-.119
Edrotes	.703	.494	.431	.100	.010	.085	.676	.457	.464	.123	.015	.085
Scarab	.652	.425	.379	.124	.015	.113	.735	.541	.677	-.101	.010	-.081
Ceramb	.606	.367	.611	.047	.002	-.005	.586	.344	.621	.061	.004	.144
CALPAR	.441	.195	.181	.021	0	.107	.369	.137	.320	.402	.162	.251
Dermap	.375	.140	.526	-.089	.008	-.047	.588	.346	.680	.012	0	.032
Orthop	.494	.244	.399	.029	.001	.109	.485	.236	.484	-.199	.039	-.097
Solifug	.267	.071	.153	.211	.045	.266	-.091	.008	-.113	-.190	.036	-.210
Scorp	.169	.028	.218	.272	.074	.105	-.106	.011	-.081	.159	.025	.056
Spider	.418	.175	.451	-.074	.006	.057	.610	.372	.501	.135	.018	.062
PSPM	-.604	.365	-.464	.409	.167	.242	-.499	.249	-.370	-.581	.337	-.366
DPM	.776	.603	.609	-.259	.067	-.173	.089	.008	.117	.784	.615	.601
MKR	-.632	.399	-.390	-.559	.313	-.329	-.643	.414	-.602	.173	.030	.035
HM	.534	.286	.466	-.190	.036	-.137	.611	.374	.526	.260	.068	.286
WR	-.795	.631	-.667	.275	.076	.236	-.379	.144	-.310	-.338	.114	-.278
CM	.121	.015	.091	.127	.016	.157	-.100	.010	-.050	.177	.031	.083
Gopher	.453	.205	.324	.447	.200	.309	.193	.037	.272	.159	.025	.123
Lizard	-.046	.002	-.113	.339	.115	.196	-.091	.008	-.124	-.110	.012	-.124
SNS	-.048	.002	-.074	.150	.023	.099	-.169	.029	-.074	-.063	.004	-.107
Bird	.502	.252	.377	-.063	.004	-.011	.201	.041	.221	.280	.078	.184

Reproductive Success

Reproductive success and vertebrate prey mass estimates are summarized in Table 2. For the four years that both DHS and East Valley were monitored simultaneously (2015, 2017-2019), DHS consistently produced higher mean owlets per nest, although only one breeding nest was monitored in DHS in 2015 so drawing conclusions for this year should be avoided. Our test results revealed significant differences between the number of owlets produced at DHS versus East Valley in 2019 (Mann-Whitney U , $U = 2$, $N_{DHS} = N_{EV} = 5$, $p = .05$ two-tailed) and for 2017-2019 combined (Mann-Whitney U , $U = 22$, $N_{DHS} = 11$, $N_{EV} = 13$, $.001 < p < .005$ two-tailed). However, test results did not reveal a significant difference between DHS and East Valley breeding success in 2018 (Mann-Whitney U , $U = 7$, $N_{DHS} = 4$, $N_{EV} = 5$, $p > .20$ two-tailed). We also tested the difference between reproductive success of DHS and Sky Valley nests in 2020, revealing no significant difference (Mann-Whitney U , $U = 3.5$, $N_{DHS} = 4$, $N_{SV} = 3$, $p > .20$ two-

tailed). We did not test the significance of difference between DHS and Sky Valley in 2019 because we collected usable data for only one breeding nest from Sky Valley.

*Table 2: Results of our BUOW monitoring efforts for 2015 and 2017-2020. Blank cells indicate that data was either not taken or deemed not reliable. Mean Owlets Observed was calculated using only breeding nests. NC = No Camera; N+ = may be more owlets present; *Mean was calculated including N+ values, N/A indicates mean was not calculated due to too few samples.*

Burrow ID	Study Year	Location	Max Owlets Observed	Total Prey Mass (g)	% Verts by Mass
DHS-1	2020	DHS	6	1175	96
DHS-2	2020	DHS	7	1413	93
DHS-3	2020	DHS	7	1438	85
DHS-7	2020	DHS	7	846	93
SV-4	2020	Sky Valley	10	1042	95
SV-5	2020	Sky Valley	6	1692	94
SV-2	2020	Sky Valley	0	858	95
SV-3	2020	Sky Valley	8	1758	92
DHS Mean	2020	DHS	6.7	1218	91.75
Sky Valley Mean	2020	Sky Valley	8	1337	94
DHS-1	2019	DHS	8	2297	89
DHS-2	2019	DHS	7	1827	92
DHS-3	2019	DHS	0	494	68
DHS-5	2019	DHS	6	1553	85
DHS-6	2019	DHS	10	1105	96
DHS-7	2019	DHS	8	3327	95
EV-2	2019	East Valley	6	2840	89
EV-3	2019	East Valley	4	1893	88
EV-7	2019	East Valley	2	377	50
EV-9	2019	East Valley	3	485	83
EV-10	2019	East Valley	7	1106	84
SV-1	2019	Sky Valley	6	267	89
SV-2	2019	Sky Valley	0	400	94
SV-3	2019	Sky Valley	NC	760	58
DHS Mean	2019	DHS	7.8	1767	87.5
East Valley Mean	2019	East Valley	4.4	1340	78.8
Sky Valley Mean	2019	Sky Valley	N/A	475	80.3
DHS-1	2018	DHS	5	1580	75
DHS-2	2018	DHS	8	1460	86
DHS-3	2018	DHS	6+		
DHS-4	2018	DHS	3+	382	58
DHS-5	2018	DHS	5	1105	80
EV-1	2018	East Valley	5	1104	33
EV-2	2018	East Valley	4	1960	80
EV-3	2018	East Valley	2	1135	72
EV-4	2018	East Valley	0	437	65
EV-5	2018	East Valley	NC	313	41
EV-6	2018	East Valley	7		
EV-7	2018	East Valley	0	254	57
EV-8	2018	East Valley	NC	454	81
DHS Mean	2018	DHS	5.4*	1131	74.7
East Valley Mean	2018	East Valley	4.5	808	61.2
DHS-A	2017	DHS	9		
DHS-B	2017	DHS	8		
DHS-C	2017	DHS	8		
DHS-D	2017	DHS	NC		
DHS-E	2017	DHS	NC		
DHS-F	2017	DHS	NC		
DHSXX	2017	DHS	NC		
	2017	East Valley	3		
	2017	East Valley	5		
DHS Mean	2017	DHS	8.3		
East Valley Mean	2017	East Valley	N/A		
	2015	DHS	0		
	2015	DHS	0		
	2015	DHS	3		
	2015	East Valley	1		
	2015	East Valley	2		
	2015	East Valley	2		
	2015	East Valley	0		
	2015	East Valley	4+		
DHS Mean	2015	DHS	N/A		
East Valley Mean	2015	East Valley	2.3*		

We found no significant change between the number of owlets produced at both DHS for 2018 to 2019 (Mann-Whitney U , $U = 3$, $N_{2018} = 3$, $N_{2019} = 5$, $p > .20$ two-tailed) or 2019 to 2020 (Mann-Whitney U , $U = 5$, $N_{2019} = 5$, $N_{2020} = 4$, $p > .20$ two-tailed) and at East Valley from 2018 to 2019 (Mann-Whitney U , $U = 8.5$, $N_{2018} = 4$, $N_{2019} = 5$, $p > .20$ two-tailed). We did not test the change between 2017 to 2018 at DHS or East Valley due to too few breeding nests.

Annual rainfall is a common predictor of reproductive success for many desert species. As such, we investigated the possible correlation between annual rainfall and owlet production (Figure 3). The only location that showed significant correlation was DHS (Pearson's Correlation, $N = 15$, $r = 0.602$, $p = 0.018$, using rainfall measurements from WWFP) for combined 2015, 2017-2020 data from individual nests. We did not include Sky Valley nests in this analysis due to a lack of a sufficient number of samples.

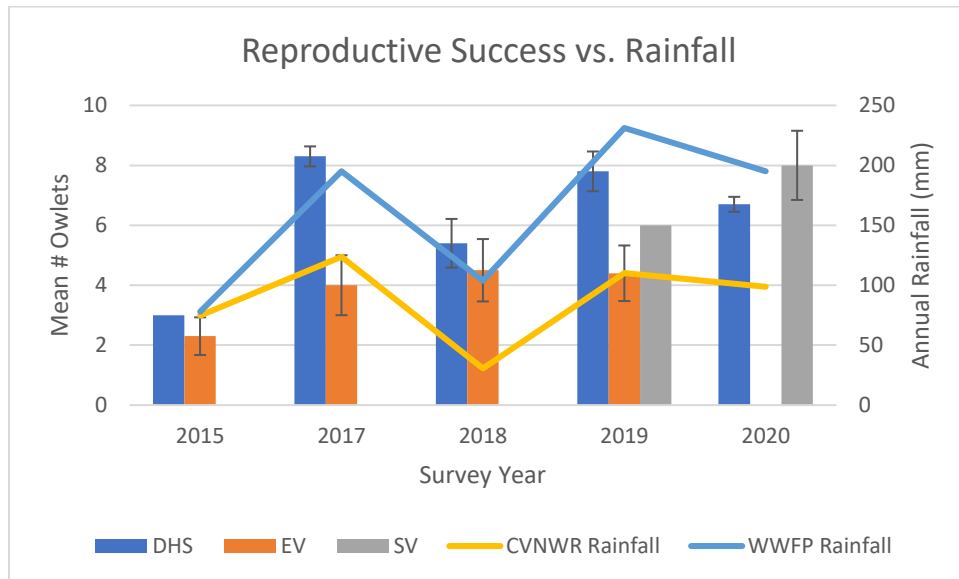


Figure 3: Mean BUOW reproductive success by survey year and location. Error bars represent standard error.

To test our hypothesis that differing ratios of vertebrates to invertebrates in the owls' diet effects breeding success, for each breeding nest we tested the estimated percent of the total prey mass that was composed of vertebrates against the maximum number of owlets produced. For all nests (DHS, East Valley, and Sky Valley) monitored between 2018 to 2020, we observed a positive correlation (Pearson's Correlation, $N = 23$, $r = .580$, $p = .004$) between percent vertebrates by mass in diet and owlets produced. Interestingly, when we tested locations individually, the correlation was much lower for East Valley (Pearson's Correlation, $N = 8$, $r = .243$, $p = .58$), and higher for DHS (Pearson's Correlation, $N = 12$, $r = .640$, $p = .025$). We did not test Sky Valley nests individually due to a lack of a sufficient number of sample nests.

We did not observe any significant changes in the percent vertebrates by mass in diet at breeding nests only in DHS from 2018 to 2019 (Mann-Whitney U , $U = 1$, $N_{2018} = 3$, $N_{2019} = 5$, p

= .10 two-tailed) and 2019 to 2020 (Mann-Whitney U , $U = 9$, $N_{2019} = 5$, $N_{2020} = 4$, $p > .20$ two-tailed) and East Valley from 2018 to 2019 (Mann-Whitney U , $U = 2$, $N_{2018} = 3$, $N_{2019} = 5$, $p = .20$ two-tailed). However, similar to our results comparing breeding success between-years, the percentage of vertebrates, by mass, consumed by owls at breeding nests at DHS and East Valley combined significantly increased from 2018 to 2019 (Mann-Whitney U , $U = 8$, $N_{2018} = 6$, $N_{2019} = 10$, $p = .02$ two-tailed). When including non-breeding nests along with breeding nests, the results are the same for DHS 2018-2019 (Mann-Whitney U , $U = 4$, $N_{2018} = 4$, $N_{2019} = 6$, $.20 > p > .10$ two-tailed), DHS 2019-2020 (Mann-Whitney U , $U = 9$, $N_{2019} = 6$, $N_{2020} = 4$, $p > .20$ two-tailed). However, we observed a significant increase in vertebrate consumption in East Valley from 2018 to 2019 (Mann-Whitney U , $U = 5$, $N_{2018} = 7$, $N_{2019} = 5$, $p = .05$ two-tailed). DHS and East Valley combined still showed a significant increase in vertebrate consumption from 2018 to 2019 (Mann-Whitney U , $U = 14$, $N_{2018} = 10$, $N_{2019} = 10$, $.01 > p > .005$ two-tailed)

Nest Visitors

2020 saw a decrease in raven visitations to nests in DHS compared to the last two years (Table 3), with an average of 4.5 raven visitations per burrow over the monitoring period, compared to 18.2 and 13 in 2018 and 2019, respectively. However, we recorded one instance of a raven taking an owlet at DHS-224-566 in 2020. It is unknown whether this raven preyed on a live owlet or whether it was scavenging a corpse. We observed only a small number of ravens in Sky Valley in both 2019 and 2020, however the nests at this site were commonly visited by kit foxes (*Vulpes macrotis*, data not shown). We did not observe any instances of predation or nest destruction by kit foxes. Roadrunners were still fairly common in DHS, but coyotes were much rarer, with only one observation across all DHS burrows in 2020. Visitation by desert cottontail rabbits (*Sylvilagus audubonii*) and black-tailed jackrabbits (*Lepus californicus*) in DHS appear to be increasing, with an average of 55.5 visitations per nest at DHS over the monitoring period in 2020. We observed an average of 7.6 and 27.5 rabbit visitations in 2018 and 2019, respectively. We also observed one instance in DHS of a round-tailed ground squirrel attacking an owl (Figure 4). The owl did not appear to be seriously injured.



Figure 4: round-tailed ground squirrel seemingly attacking a young owl in DHS.

Table 3: list of non-owl visitors and predation events observed at BUOW nests from 2017 to 2020. Animals that we decided were not a threat to owl health (such as mice and lizards) were not included, except for rabbits (cottontail and black-tailed jackrabbits) due to their extremely high visitation rate.

Burrow	Days observed by camera	Raven	Roadrunner	Coyote	Human	Rabbit	Predation Events
2020							
DHS-1	4/16-6/23	4	9	0	0	119	1 (Raven)
DHS-2	4/16-6/16	6	5	0	0	21	
DHS-3	4/16-6/16	2	6	1	0	35	
DHS-7	4/16-6/16	6	0	0	2	47	
SV-5	4/27-6/19	1	0	0	0	1	
SV-4	4/27-6/19	0	0	0	0	0	
SV-2	4/16-6/19	0	0	0	0	37	
SV-3	4/16-6/4	3	0	0	0	0	
2019							
DHS-3	4/26-6/26	14	4	1	2	49	1 (Raven)
DHS-5	4/26-6/26	14	4	4	0	22	
DHS-7	4/26-6/19	7	1	3	9	40	
DHS-6	4/26-5/12	7	0	0	0	0	
DHS-1	5/2-6/26	33	12	6	0	41	1 (Raven)
DHS-2	5/12-6/26	3	2	2	0	13	
SV-2	5/16-6/21	0	0	0	0	0	
SV-1	5/20-6/12	0	0	0	0	3	
EV-2	4/24-5/20	17	1	10	0	2	
EV-3	4/25-5/20	0	0	0	0	4	
EV-7	5/20-6/26	0	6	2	0	11	
EV-9	5/20-6/26	0	1	1	0	3	
EV-10	5/20-6/26	0	0	3	0	0	
2018							
DHS-2	5/21-6/27	43	2	0	0	11	1 (Coyote)
DHS-1	6/4-7/6	10	2	0	0	6	
DHS-5	5/22-6/20	9	2	0	0	1	
DHS-3	5/18-6/26	26	2	1	0	12	
DHS-4	5/23-6/25	3	0	0	0	8	
EV-2	5/11-5/17, 5/23-7/27	7	5	3	0	29	
EV-1	5/23-6/25	2	4	1	2	0	
EV-3	5/11-5/17, 5/23-6/4, 6/7-6/29	0	0	0	0	14	
EV-7	5/18-6/29	0	0	0	0	4	
EV-6	5/17-6/5	0	3	1	0	0	
EV-4	5/17-6/4	2	0	0	1	0	
2017							
DHS-A	4/17-7/31	3	2	5	0		
DHS-B	4/17-5/7, 5/15-7/31	5	0	9	0		
DHS-C	4/24-5/7, 5/15-6/25, 7/03-7/31	2	1	5	0		
CVSD10	5/8-5/22, 5/29-7/31	1	2	0	0		
CVSD13	5/8-7/31	1	2	1	2		

DISCUSSION

Our monitoring efforts of burrowing owls throughout the Coachella Valley highlights both their capacity to sustain populations within some human-modified landscapes, as well as the variety of unique threats it faces as a result of continued human encroachment. In general, owls in the relatively natural habitats at our DHS and Sky Valley sites showed higher reproductive success than their East Valley counterparts. However, the owls occupying natural landscapes also showed an increased sensitivity to fluctuations in rainfall. This sensitivity may prove to be a critical factor in the continued survival of populations in natural arid habitats as temperature increase and the frequency and intensity of droughts are expected to increase as a symptom of climate change (Seager et al. 2007, Gonzales et al. 2018, Woodhouse et al. 2010). This differential sensitivity to rainfall is most likely explained by its impact on the owls' prey base. In an open desert environment seasonal rainfall is integral in determining yearly primary productivity and hence the diversity and abundance of species that rely on annual foliage and seed production (Whitford 1976, Hereford et al. 2006). Drought years tend to result in relatively little seed germination and/or increased plant mortality and therefore negatively affect the base resource for the mammals and insects that burrowing owls rely on as a food source (although other biotic factors also contribute, e.g. Brown and Ernest 2002). However, in wet years, these natural habitats have the ability to exhibit increases in plant biomass through the germination of a wide variety of annual plant species present in the seed bank and subsequently revitalize prey abundance and diversity (Beatley 1969, 1974, Clauss and Venable 2000, Ernest et al. 2000). As we demonstrated, owlet production is significantly correlated to the percentage of vertebrate mass in the diet at DHS. Nests in DHS and Sky Valley consumed the most vertebrates and likewise produced the most owlets on average. However, in the East Valley, reproductive success was not significantly correlated to the amount of vertebrate mass consumed. This indicates that there are other factors influencing reproductive success in the East Valley.

Conversely, in the East Valley, perennial water sources are regularly available through agricultural irrigation and wastewater channels. This may allow for a stable prey availability for many species regardless of precipitation, especially those that thrive in agricultural landscapes such as many ruderal or pest arthropods. The fact that there was no statistical difference between reproduction in DHS and East Valley in 2018, a dry year, supports this hypothesis. As shown in Figure 3, rainfall was low in 2018, which correlated with reproductive success in DHS decreasing to near the level of East Valley. However, East Valley showed a slight increase in reproductive success from the previous year, which may be due to better prey abundance stability. Based on the results of our pellet dissections, ruderal insects such as dermapterans, orthopterans, and scarab beetles are an important component in the diets of East Valley owls, as are predatory arthropods such as spiders, scorpions, and solifuges that likely benefit from these insect populations. However, the abundance of high-quality prey items such as mammals may ultimately be negatively impacted by such vast urbanization (McKinney 2008). Mammals that thrive in agricultural landscapes such as Botta's pocket gopher and harvest mice are more common prey items in the East Valley, along with some apparent generalists such as desert pocket mice. Despite this, the overall mean vertebrate consumption in the East Valley is lower

relative to DHS and Sky Valley (although not statistically significant). This indicates that prey species such as gophers and harvest mice may exist at lower abundances and/or be more difficult for owls to hunt. Halen and Rosenberg (2013) demonstrated that food limitation may not affect clutch size but has an important effect on post-hatch owlet survival, particularly among last-hatched owlets. York et al. (2002) also speculate that their observed lack of rodents in BUOW stomach contents from Imperial Valley may be correlated to their low reproductive success. While we rarely observed owlet mortality, it is conceivable that owlets perished from food limitation before they were old enough to emerge from the burrow where they could be observed by our cameras.

Predation on BUOWs was not common. DHS was the only site where we observed predation. Three of the four total predations events here were carried out by ravens which we speculate are at a higher concentration in this area due to an attraction to the frequent, and often large-scale, homeless settlements and illegal dumping present here. This emphasizes the effect that nearby human influences can have on otherwise protected and fairly natural habitats. Neither Sky Valley nor the East Valley nests suffered such regular harassment by ravens. However, nests in the East Valley appear to be very susceptible to unintentional human disturbance via the maintenance practices present along the storm channel. We noted a few instances of heavy machinery collapsing burrows as they drove along the top of the channel embankments. Also, some burrows along Waste Way 2 along 52nd Avenue appear to have been destroyed or buried during riparian vegetation removal. These incidents did not occur during peak breeding season and it is not known if any owls were harmed.

Ultimately, our results reveal that in the Coachella Valley, burrowing owls occupying natural desert landscapes enjoy higher quality, and likely quantity, of prey items during times of good precipitation. However, these owls may be more sensitive to drought as prey populations decline in response. This lack of prey stability may be a cause for concern for conservation of burrowing owls in natural desert habitats, especially in the face of increasing frequency and intensity of droughts. Also, owls in DHS appear to be at higher risk of predation. On the other hand, the sources of water present in the agricultural and urban matrix of the East Valley may offer a buffer against prey population fluctuations for the owls here, and a subsequent reduction in the severity of reproductive success fluctuations. The quality of prey seems to be lesser (fewer vertebrates), however, resulting in overall lower reproductive success, and nests are more likely to be destroyed by human activity such as regular channel embankment maintenance. Catlin and Rosenberg (2006) highlight the potential for owl mortality and nest destruction by such maintenance in the Imperial Valley and suggest the installation of well-marked artificial burrows to mitigate losses. It would be useful to investigate whether the East Valley population is self-sustaining, or if it is a sink for surrounding populations occupying natural habitat. If the proposed buffering effects of the East Valley are correct, this area, and others like it, may have the critical potential to serve as a refuge for burrowing owls if future drought severely impacts populations in natural landscapes.

Our monitoring methods result in highly detailed data involving three critical aspects of burrowing owl survival – diet, nest disturbances, and subsequent reproductive success. However,

the level of detail we achieve here results in the necessity of larger investment of manpower into a relatively small number of nests. The complexity of our monitoring efforts also means that some of our early efforts required subsequent revising to ensure the best possible data is collected. Thus, some early results are not directly comparable to the previous three years of surveying. Despite this, we believe that our methods offer valuable insight into several important aspects of burrowing owl ecology in the Coachella Valley. A narrower approach in the future (such as recording only reproductive success), may help to increase the overall sample size and statistical power related to specific issues. For instance, developing a survey protocol that employs regular point surveys on a much larger set of nests will provide better information regarding reproductive success over a wide range which can be used to help determine what populations, if any, serve as sinks.

Management Recommendations

In DHS, it is important to keep raven harassment at a minimum. In addition to predation, we have also recorded ravens steal food from the owls' caches. Even if they do not predate an owl or steal food, their presence is likely stressful. If correct, our connection to raven density and food source availability from illegal dumping and homeless encampments must be considered in this area. The levels of ravens in 2020 was much lower than previous years, suggesting that the area may be on a healthy trajectory, but this must be maintained by ensuring sources of attraction for ravens are not reestablished.

Owls in the East Valley appear to be threatened by maintenance activities along the wastewater channels. Better marking of burrow locations, communication with maintenance crews, and more careful application of some maintenance methods (such as mulching of riparian vegetation and deposition onto surrounding embankments) will benefit this population of owls.

Future research may include investigation of factors such as rates of vehicle collisions or effects of pesticides, both of which may be important in the East Valley.

Since BUOW seemingly rely on small mammals to start burrows, monitoring and population trends of round-tailed ground squirrels and California ground squirrels in the areas owls occupy may also help elucidate the owls' population trajectory.

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Palm Springs Pocket Mouse

The Palm Springs Little Pocket Mouse (*Perognathus longimembris bangsi*, PSPM) is a species of special concern in the state of California as well as one of the species covered under the Coachella Valley Multiple Species Habitat Conservation Plan. We monitor this species annually as part of our aeolian vertebrate tracking efforts (see UCR-CCB 2020 Aeolian Sand Species Monitoring Report for survey methodology). Surveying by tracking offers several advantages: since this is a nocturnal species, biologists do not have to work during late hours of the night; the mice do not have to be handled, which prevents the chance of contracting or spreading harmful diseases; and traps are not required, which eliminates stress to the animals. However, tracking requires the presence of fine, loose substrate such as dune sand or blowsand which is regularly “cleaned” by strong winds. Such conditions are present across only a portion of the PSPM’s potential habitat, which occurs throughout most of the Coachella Valley, from the San Geronio Pass to Borrego Valley (UCR CCB 2009, Dodd 1996).

As part of our burrowing owl (*Athene cunicularia*, BUOW) monitoring efforts, we collect regurgitated pellets and identify the owls’ prey items. A significant portion of the owls’ diet is composed of small mammals, including PSPM. Therefore, we determined it is possible to supplement our investigation of PSPM population dynamics to include areas other than aeolian systems via pellet dissections from owls that do not hunt in aeolian habitats.

Figure 1 shows results from our 2019 and 2020 PSPM tracking efforts across our aeolian community plots. Interestingly, while PSPM density has remained fairly stable at our CVNWR, except for increases on the active dunes, density has dramatically increased at the Whitewater Floodplain Preserve (WWFP). It has also roughly doubled at Kim Nicol Trail (KN) and Willow Hole (MH 19-29) yet dropped to zero at Stebbins Dune (SD). This is likely the cumulative result of three wet years in close proximity (2017, 2019, and 2020). If the winter of 2020/2021 is fairly wet, we might expect to see either further increases in PSPM density or a plateauing due to the population reaching carrying capacity.

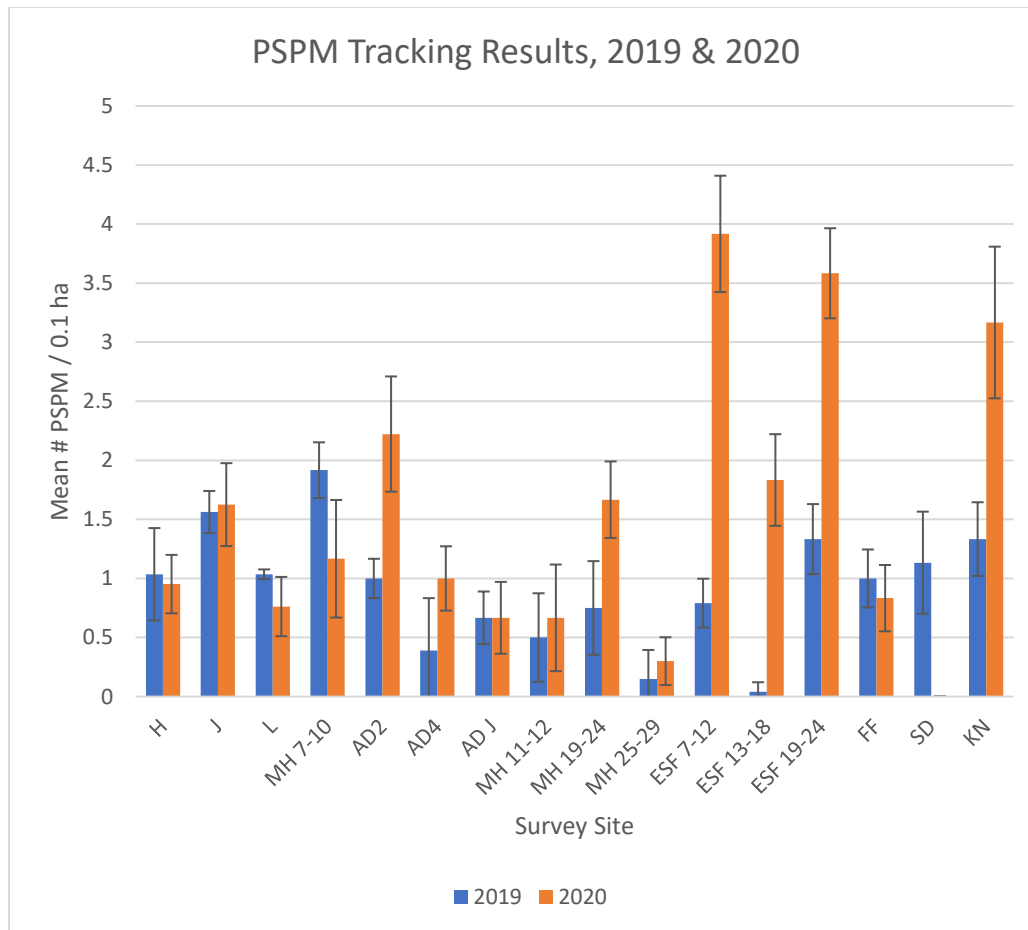


Figure 1: Mean PSPM density (PSPM/0.1ha) across all aeolian monitoring plots for 2019 and 2020. Error bars indicate standard error.

Figure 2 show our combined PSPM monitoring results for all aeolian tracking plots since 2008, plotted alongside annual rainfall. Species that live in arid environments such as the Coachella Valley are usually very sensitive to changes in precipitation. We hypothesize that PSPM respond similarly to rainfall, with decreases in population corresponding to decreases in annual precipitation. Figure 2a displays rainfall amounts per monitoring year, and there is no correlation between PSPM density and annual rainfall at the Whitewater Floodplain Preserve or the Coachella Valley National Wildlife Refuge (CVNWR) (Pearson's Correlation, $r=.36$, $p=.25$; $r=.2$, $p=.71$, respectively). However, offsetting the rainfall by one year into the future, as shown in Figure 2b, reveals a better correlation with PSPM density. This can be explained by a "lag time" that some animals exhibit – in essence, populations do not rebound instantly as soon as precipitation increases. Instead, a rebound is observed only after enough time has passed for the effects of the increased rainfall (more plant food growing, better breeding season, etc.) to taken place. In the case of small mammals like the PSPM, the effects of a good breeding season and an increase in population may not be apparent until the next survey effort during the following year. Figure 2 lends evidence to this hypothesis, showing rainfall at both WWFP and CVNWR

tracking closely with PSPM density from 2015 to 2020 (Pearson's Correlation, $r=.86$, $p=.03$ for both WWFP and CVNWR). However, no correlation is observed from 2008 to 2013 (Pearson's Correlation, $r=-.02$, $p=.97$ (WWFP); $r=.29$, $p=.57$ (CVNWR)) or from 2008 to 2020 ($r=.37$, $p=.24$ (WWFP); $r=.46$, $p=.13$ (CVNWR)). The cause of this lack of correlation with rainfall between 2008 and 2013 is unknown, but likely involves the effects of the preceding drought years and competition with other species.

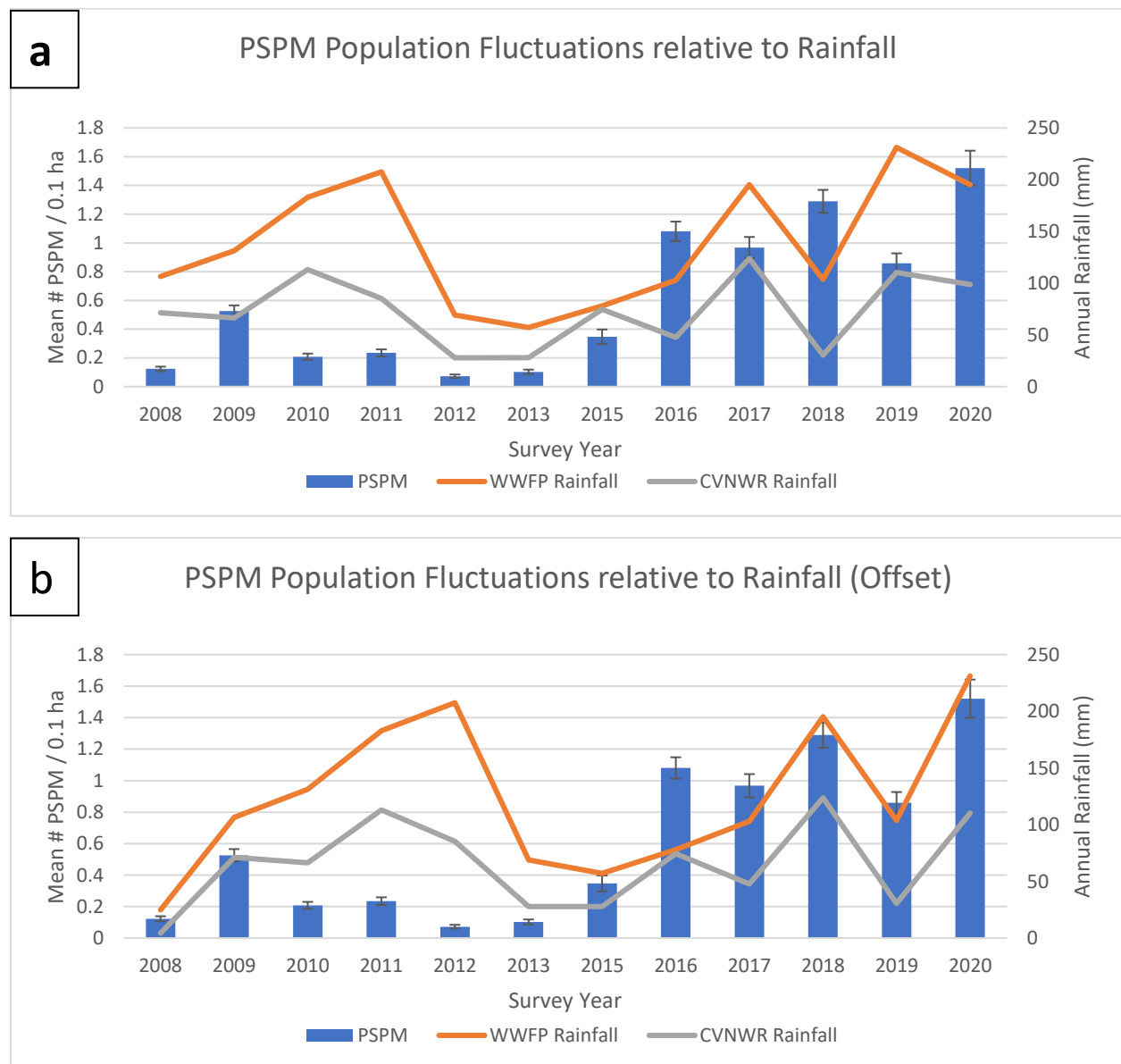


Figure 2: Mean PSPM densities for all aeolian plots combined for 2008 to 2020 (excluding 2014). The orange line indicates annual precipitation amounts taken at the Whitewater Floodplain Preserve (WWFP). The gray bar indicates annual precipitation amounts taken at the Coachella Valley National Wildlife Refuge (CVNWR). Figure 2a shows precipitation in relation

to the year the measurements were taken; Figure 2b shows precipitation offset by one year into the future.

In DHS, the pattern of PSPM abundance, by total mass, in the diets of our monitored burrowing owls appears to coincide with the results of our tracking efforts (Figure 3). However, the diets of owls in East Valley and Sky Valley on average contained much lower amounts of PSPM. The habitat surrounding the East Valley owl nests is largely anthropogenic, which would explain the relative lack of PSPM there. However, the Sky Valley nests occupy the most pristine habitat out of the three study locations. There are perhaps three hypotheses that could explain this: habitat suitability, hunting preference, and competition. The upland areas within the Sky Valley owls' hunting range contains much less fine sand than DHS, and instead is largely alluvial rock and gravel, which may not be as suitable for large PSPM populations. Similarly, the habitat conditions in this area may favor desert pocket mice (*Chaetodipus penicillatus*) and Merriam's kangaroo rats (*Dipodomys merriami*), which would exert a competitive pressure on the PSPM because these species all rely on the same food source (seeds). The owls at Sky Valley usually have a much higher percentage of desert pocket mice and Merriam's kangaroo rats in their diet. This may simply be a reflection of the lack of PSPM in the area, or it could be due to preferential hunting – even if PSPM densities are high, if desert pocket mice and Merriam's kangaroo rat populations are also sufficiently high, the owls may exhibit a preference for these two species versus PSPM. PSPM are significantly smaller than desert pocket mice and kangaroo rats, so it is logically more energetically efficient to hunt these species instead.

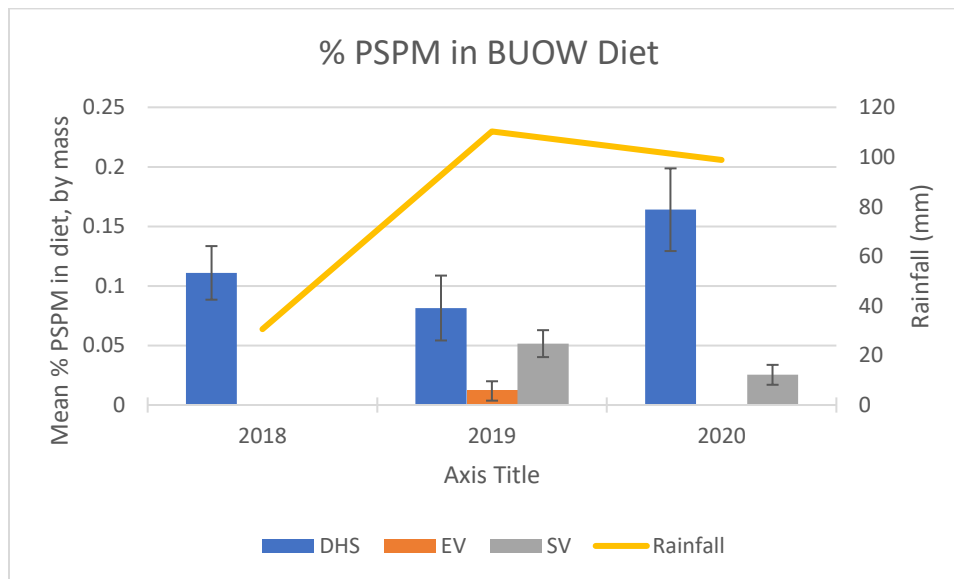


Figure 3: Percent PSPM, by mass, in the diet of burrowing owls monitored from 2018 to 2020. In 2018, we identified no PSPM in the diet of East Valley owls, and we did not monitor Sky Valley owls. In 2020, we did not monitor East Valley owls.

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Appendix VII-
Survey Results for
Orocopia Sage (*Salvia greatae*) &
Mecca Aster (*Xylorhiza cognata*)

Coachella Valley Multiple Species Habitat Conservation Plan
Survey Results for
Orocopia Sage (*Salvia greatae*) & Mecca Aster
(*Xylorhiza cognata*)



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Introduction

This report documents the current status of Orocopia sage (*Salvia greatae*), and Mecca aster (*Xylorhiza cognata*), two plant species whose ranges are largely confined to the Coachella Valley and among the 27 species covered under the Coachella Valley Multiple Species Habitat Conservation Plan (CVMSHCP). While neither species is threatened by urbanization, generally occurring in remote and conserved lands, other threats ranging from climate change, invasive species, off-road vehicle recreation, and sand and gravel mining represent potential challenges to the persistence and so successful conservation and protection of these two plants. Identifying the extent to which such threats may be impacting these plants can inform land management priorities aimed at ensuring the on-going persistence of species protected under the CVMSHCP.

Orocopia sage is a shrub endemic to Orocopia and Chocolate mountains within the Colorado Desert of Riverside and Imperial counties, California. Although it has no status under the State of California or Federal Endangered Species Acts, the California Native Plant Society (CNPS) has designated this species with a California Rare Plant Rank categorization of 1B.3 (threatened or endangered in California and elsewhere, with < 20% of occurrences threatened / low degree and immediacy of threat or no current threats known). The described habitat of *S. greatae* includes alluvial fans, slopes, and washes between 30-450m, although CNPS reports it as occurring in a larger elevational range, from 40m below sea level up to 825m (Averett 2012, CNPS 2020).

Mecca aster is also a perennial shrub, and is endemic to the Indio and Mecca Hills in Riverside County, California. CNPS categorizes this species as California Rare Plant Rank 1B.2 (threatened or endangered in California and elsewhere, with 20-80% of occurrences threatened / moderate degree and immediacy of threat; CNPS 2020). Mecca aster are typically found on steep canyon slopes, especially near the toe of those slopes as they interface with canyon bottoms, at elevations between 20-300m and is associated with Sonoran Desert scrub (Keil 2012, CNPS 2020).

In 2002, UC Riverside (UCR) compiled a database of historic occurrence records for all five plant species covered under the CVMSHCP (Allen et al. 2005). The UCR research team mined data querying various herbaria and museums, identified points georeferenced precisely enough for inclusion, and visited occurrence locations on public land for each species; documenting the existing populations using 500m² vegetation relevés (Allen et al. 2005). More recently, in 2014, a UCR research team selected *S. greatae* and *X. cognata* survey sites by identifying groups of occurrence records existing within conservation areas and within modelled areas of potentially suitable habitat. Each site consisted of a combined four survey points, researchers employed the CNPS-CDFW Combined Vegetation Rapid Assessment and Relevé protocol. A survey point was one 10X100m (1000m²) relevé transect that documented habitat attributes such as the vegetation community, local disturbances, and presence and extent of invasive plants (Buck-

Diaz and Evans 2011). At each survey point researchers counted *S. greatae* and *X. cognata* individuals and documented life stages of *X. cognata* individuals.

In the 2002 survey effort, *S. greatae* populations were surveyed at seven of the 10 unique public land records. The recorded attributes at each population varied but averaged over 200 individuals per hectare at each location and, several survey points appeared to be sampling continuous population patches, covering large areas (Allen et al. 2005). The research team noted that *S. greatae* was present and numerous over large areas within its range, and hypothesized that additional populations may exist at upper elevation sites in the Orocopia Mountains. In 2014 the UCR research team revisited verified locations from 2002 as well as established additional sites. The team visited four of the seven occupied sites (as above, from 2002) and verified live populations of *S. greatae* at all four sites. Further effort was made to verify more historic points and a total of 14 sites were established, with an additional 15 additional incidental occurrences (target species observed, locations recorded, but no plots created, or surveys conducted). These incidental occurrences included undocumented populations and extensions of documented populations and were recorded to further refine the habitat suitability model (UCR 2014).

In 2002, the UCR database identified 13 unique *X. cognata* records occurring on public lands. UCR researchers surveyed each of those 13 sites and observed individuals at 12 locations and identified two new locations where the species occurred. Population sizes at all occupied locations varied from six individuals to 377, averaging about 80 individuals per location. Most records for *X. cognata* were recorded in the Mecca Hills area; there was then only one record of *X. cognata* occurring on public land in the Indio Hills area. In 2014, the UCR research team resurveyed 12 of the original 13 sites and verified live populations at all but one site, consistent with 2002 findings. Along with these 12 sites, eight more sites were identified from historic public land records, for a total of 20 sites. The research team confirmed populations at 16 of the 20 sites, and documented an additional 42 incidental occurrence points of *X. cognata*. These points were either an extension of recorded populations or separate, isolated populations, and used to refine the habitat suitability model (UCR 2014). As with the 2002 assessments, *X. cognata* was primarily found in the Mecca Hills, however two of the new survey sites were established in the western Indio Hills, expanding on the single site from 2002 in the East Indio Hills (UCR 2014).

Objectives

Our overall objective for the 2019 monitoring effort was to carry out surveys for *Salvia greatae* and *Xylorhiza cognata* as required by the CVMSHCP. We conducted surveys following the guidelines and objectives outlined by the CVMSHCP, using protocols we developed in combination with the 2014 modified vegetation relevé protocol developed by the Coachella Valley Conservation Commission's Biological Working Group (BWG). The primary objectives for this monitoring effort mirrored those of 2014; for both species, we wanted to assess the

current presence, distribution, and populations demographics, as well as to document habitat attributes, identify potential stressors that may affect species persistence, such as fire, invasive plants, off-road vehicles (OHV), trampling, and altered ecosystem processes due to flood control measures, and climate change impacts.

Additional objectives included establishing a detailed and repeatable monitoring protocol that efficiently captured the above concerns, re-monitor survey sites established in 2014, further investigate known past occurrence recordings and newer documented occurrences since 2014, and further explore modeled suitable habitat for both species to find undocumented populations.

Methods

The 2014 survey effort employed a modified version of the CNPS-CDFW Combined Vegetation Rapid Assessment and Relevé protocol (Buck-Diaz and Evans 2011) to assess populations of both *Salvia greatae* and *Xylorhiza cognata*. However, due to the differences between the species' habitats, we redesigned the 2014 survey method to better capture population demographics, distribution, and density, as well as to provide specific protocols for increased repeatability. This resulted in our development of a unique monitoring protocol for each species which are individually better suited for monitoring in the varied habitats these plants occupy.

Salvia greatae

Site Selection

We conducted *Salvia greatae* surveys from October 2019 to February 2020. In 2014, researchers visited 55 relevé survey points that were clustered in approximate groups of four, equaling a total of 14 sites. To better observe the how nuanced elevation changes, topography, and habitat differences effect *S. greatae*, we broke these clusters into independent survey sites.

Independent survey sites were selected to sample different ranges in elevation of *S. greatae* habitat. We completed surveys on a total of 30 independent sites; 15 were revisits of sites established in the 2014 survey effort (representing half of the original 2014 sites), 6 were sites where *S. greatae* was identified in the Mecca Hills and Orocopia Mountains vegetation mapping effort (UCR 2015), and we established 9 of the 30 sites using searches in the field, exploring suitable habitat and using observations based on recent locality additions to California Natural Diversity Database (CNDDDB; <https://wildlife.ca.gov/data/cnddb>) and Calflora database(www.calflora.org/). *S. greatae* was present at 18 of the 30 independent survey sites (Figure 1).

Despite historical records, there have been no observations of this species within the last 20 years within the Dos Palmas basin, along Salt Creek, and along the Coachella canal. The 2014 research team surveyed these areas and verified that *S. greatae* was no longer detectable, therefore no additional surveys were conducted in those areas in 2019. It is unclear whether this species has been extirpated from these low elevation, high soil salinity sites, or whether they

were never present there and the site locations were generalized descriptions of occupied sites within 2-6 km north and east of the canal and Dos Palmas Basin. Additionally, in 2014 UCR documented the extent of a large *S. greatae* population along the Bradshaw Trail. Although we did revisit several of the absence points in this area, most of this population is located on land that is part of the Chocolate Mountains Aerial Gunnery Range (outside the boundary of the CVMSHCP), and as we did not have permission from the Department of Defense to enter this area to perform biological inventories, those sites were excluded from our 2019 surveys.

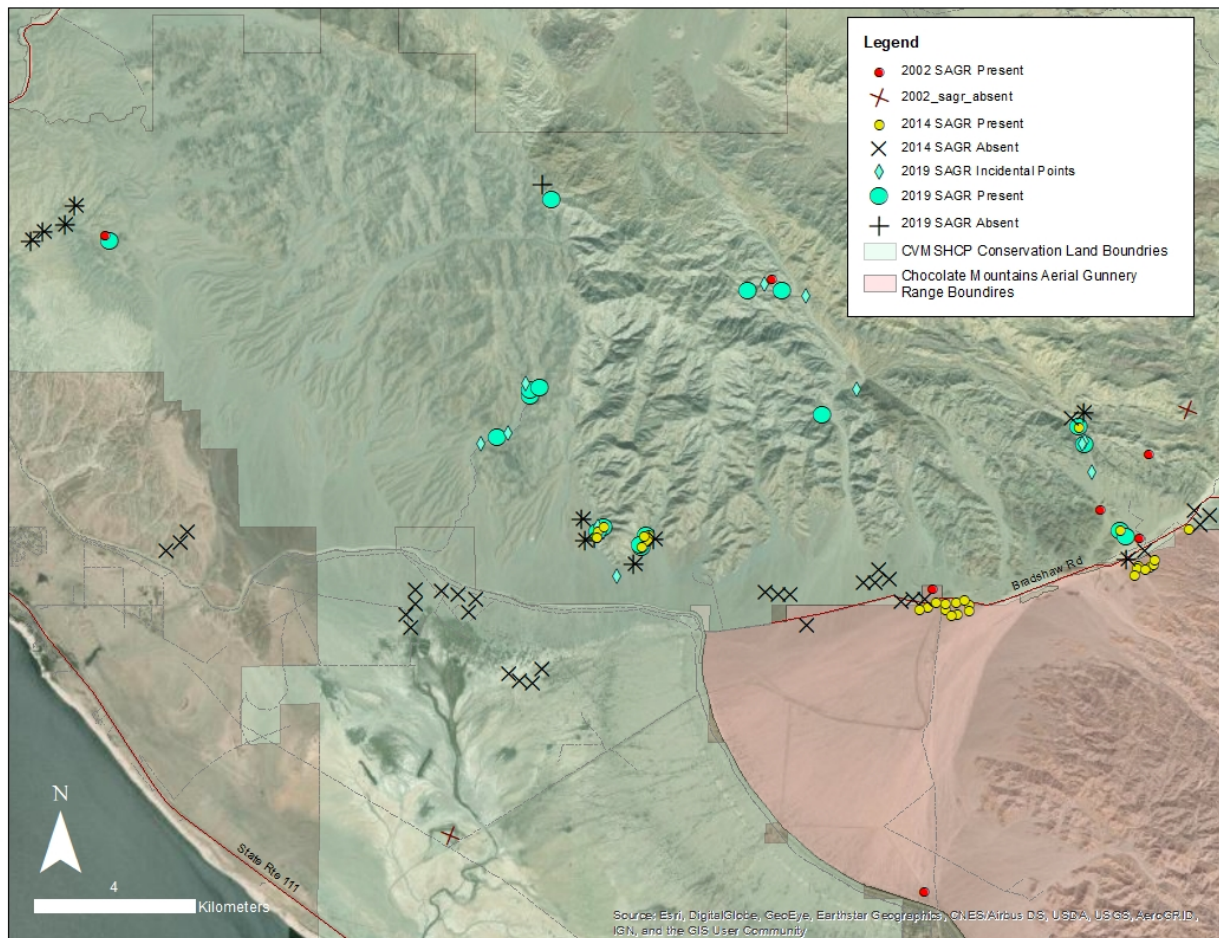


Figure 1: 2014 and 2019 *Salvia greatae* survey sites.

Protocol

We replaced the 10x100m modified relevé transect with a two part survey protocol; a 30m radius (area of 2827m²) survey, and a 20x200m (4000m²) belt transect survey (Figure 2). We developed the 30m radius survey method to get a standardized density count per area at each site. We only implemented the belt transect survey in broad bajadas to determine whether *S. greatae* is evenly distributed across the bajada, or whether it has a more restricted distributions such as occurring within braided washes throughout bajadas and against bajada walls.

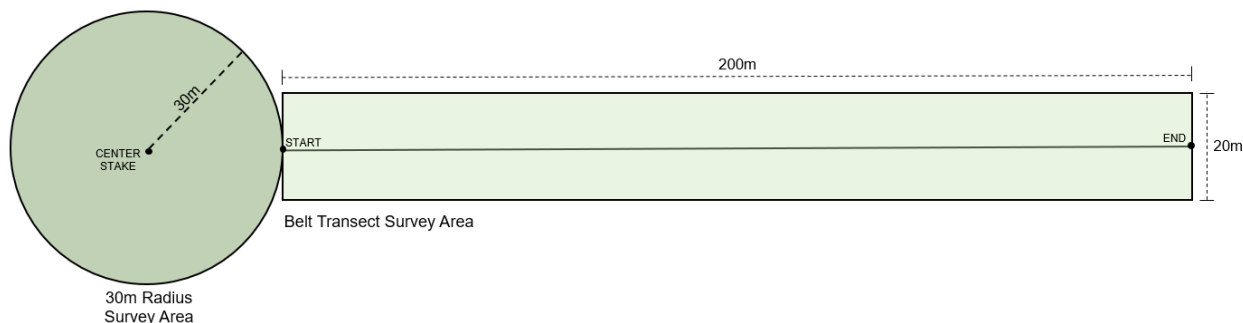


Figure 2: *Salvia greatae* 30 meter radius survey area and 20x200 meter belt transect survey area.

We accessed survey points on foot and took care to not impact the species and their habitat during monitoring. Due to the inherent GPS error from signal obstruction and multipath error canyons, we located the survey points as precisely as we could by cross referencing coordinates using a Trimble Juno 3B GPS unit (www.trimble.com/) and a Samsung Galaxy Tab Active 2 tablet (www.samsung.com/us/) running a georeferenced PDF map we created on ArcMap (ArcGIS Desktop, www.esri.com; version 10.9) on Avenza maps (www.avenza.com).

Once the location of the survey point was established, we marked the point with a 12" galvanized iron stake with a washer and numbered tag attached, which became the center point for 30m radius survey (Figure 3). At the center point we recorded UTM coordinates (NAD 83 UTM Zone 11S), the tag number, and took North, South, East and West photos. Photo numbers were recorded, and photos are archived on the CCB network server available upon request to the CVCC.



Figure 3: Center stake for *Salvia greatae* survey site ORO-22, showing *S. greatae* individuals occurring in rocky substrate.

Within the 30m radius we tallied and categorized the health of all *S. greatae* individuals (rooted inside the circle) on a five point scale, where '1' represented 0% of the total plant was alive (i.e. completely dead, <2 years), '2' was 1-24% alive, '3' was 25-49% alive, '4' was 50-74% alive, and a '5' was 75-100% alive. A distinction was made between adults and juveniles or seedlings, which were defined as being <20cm tall and have never flowered. We used both the Samsung tablet and a Hilti PD-42 laser range meter (www.hilti.com) to verify that *S. greatae* plants were within 30m of the center point. If the terrain was too fragile or rugged to access individuals, we used the laser range meter and binoculars to determine distance and health of individuals.

Specific disturbances of concern that were recorded were OHV activity, natural erosion, invasive species (species were recorded), trampling due to foot traffic or social trails, and dumping or littering. Disturbances within the 30m radius were evaluated on the following scale: light was less than 33% of the population within the plot was impacted; moderate was 33-66% impacted; and high was greater than 66% of the population within the plot was impacted by the disturbance. We also recorded the presence of other perennial plants and made notes

about significant terrain, geology, or unique attributes of the area. A copy of the datasheet we used in the field for the *S. greatae* protocol can be found in Appendix 1.

While data from 2002 and 2014 indicated that *S. greatae* could be found in clusters, our pre-survey observations revealed that additional plants were located scattered within the bajadas. To estimate density and distribution of these plants, we launched parallel line-distance transects to bisect the broad bajadas, starting on the outer perimeter of a 30m radius plot (Figure 2). We traversed these surveys on foot, as close to the transect line as possible (considering the undulating topography throughout the bajadas) two surveyors counted plants within 10m on either side of the line. We measured the distance of *S. greatae* individuals from the transect line by using a laser range meter. We recorded the UTM coordinates and health assessment rating (in respect to the above listed). The length of the transect depended on the contour-line width of the bajada but did not exceed 200m.

If *S. greatae* plants were not apparent upon arrival at the site we spent at least five minutes between two surveyors searching the surrounding area. If we did not find that there were *S. greatae* plants (live or dead) present, we documented coordinates and recorded all other site data. Incidental occurrences were noted and are included in the database submitted as part of yearly monitoring activities.

Xylorhiza cognata

Site Selection

We surveyed *X. cognata* from October to December of 2019. We kept the survey sites in clusters in accordance with the 2014 method and were able to survey 19 of the original 20 (ulti-survey) sites, with the addition of one site in the western Mecca Hills region and several survey points at previously established sites, for a total of 73 individual surveys (Figure 4). Additional occurrence locations were sourced from CNDDDB and Calflora, and were the result of our continued exploration into *X. cognata* modeled habitat.

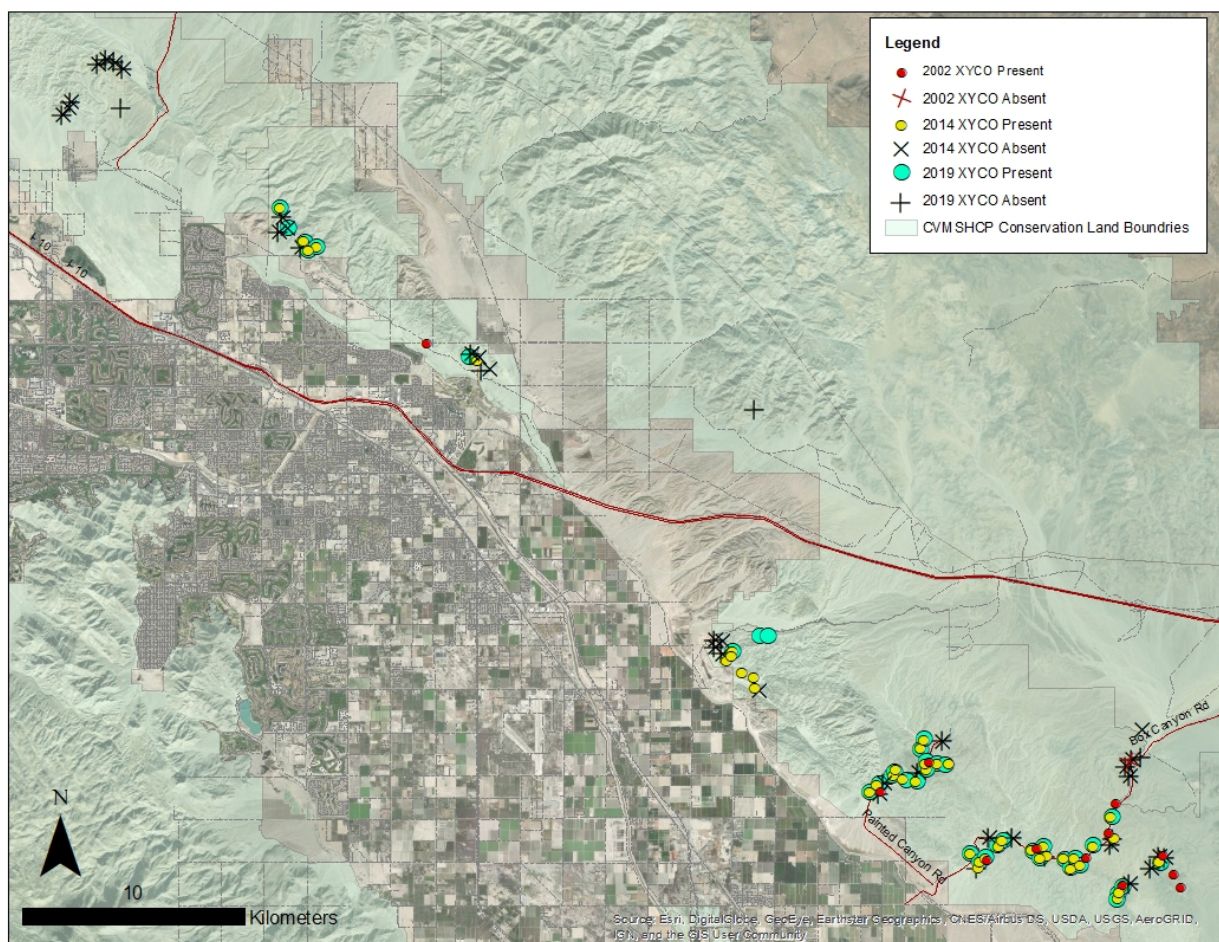


Figure 4: 2014 and 2019 *Xylorhiza cognata* survey sites.

Protocol

We altered the 10x100m relevé transect to be 5m by up to 100m; allowing the length of the transect to be determined by specific conditions. These conditions were topography and habitat; when the transect terrain became too dangerous or delicate for us to traverse (e.g. dry waterfalls, talis slopes, and loose boulders). The transect did not necessarily end at the extent of the population but ended at the extent of our ability to access the population. The defined habitat of *X. cognata* is inherently precarious for humans and we captured as much as we could without putting ourselves at risk or damaging *X. cognata* individuals or habitat. Another reason that transects were truncated was when it reached a canyon mouth and the terrain, and this the habitat, because flat and unsuitable for *X. cognata*. We changed the width of the transect belt from 10m to 5m because in some cases, the height of the canyon wall was less than 10m, so we were not be able to account for the portion of the belt that we could not see. The position of the belt (whether it was on one side or bisected by the transect line) was also determined by habitat and topography. In the original relevé transect method, the belt is bisected by the transect,

however in the case of wide canyons, this method would cause the belt to be half in the canyon wash, thus sampling unsuitable habitat.

As with *S. greatae* surveys, due to the inherent GPS error from working in canyons, we located the survey point as precisely as we could by cross referencing coordinates using a Trimble Juno 3B GPS unit and a Samsung Galaxy Tab Active 2 tablet running a georeferenced PDF map we created in ArcMap on Avenza maps.

We accessed survey points on foot and took care to not impact the species of concern and their habitat during monitoring, once the location of the survey point was established, we marked the point with a 12" galvanized iron stake with a washer and numbered tag attached, which became the start point of the transect line. We then set up the transect by running a 100m reel along the canyon bottom. At the start and end of the transect we recorded UTM coordinates, tag number, bearing from start to end of transect (acquired using a compass with declination set), total length of the transect, and took a photo of the transect line (photos are stored on the CCB server and shared with CVCC).

As we traversed the winding slot canyons with the reel we set up vertices in the transect line where the canyon turned to keep the transect in the canyon bottom and the belt within suitable habitat. At each vertex we used rocks to hold the line in place, recorded the location on the transect (in cm) where it occurred, the new bearing the transect took after the turn, and a photo of the vertex toward the transect end (Figure 6 and Figure 7). The tape was wound around rocks in some cases, and readings were taken exactly as indicated on the tape, resulting in a nominal error of likely 10-40cm per transect where this modification was employed, unlikely to impact population estimates.

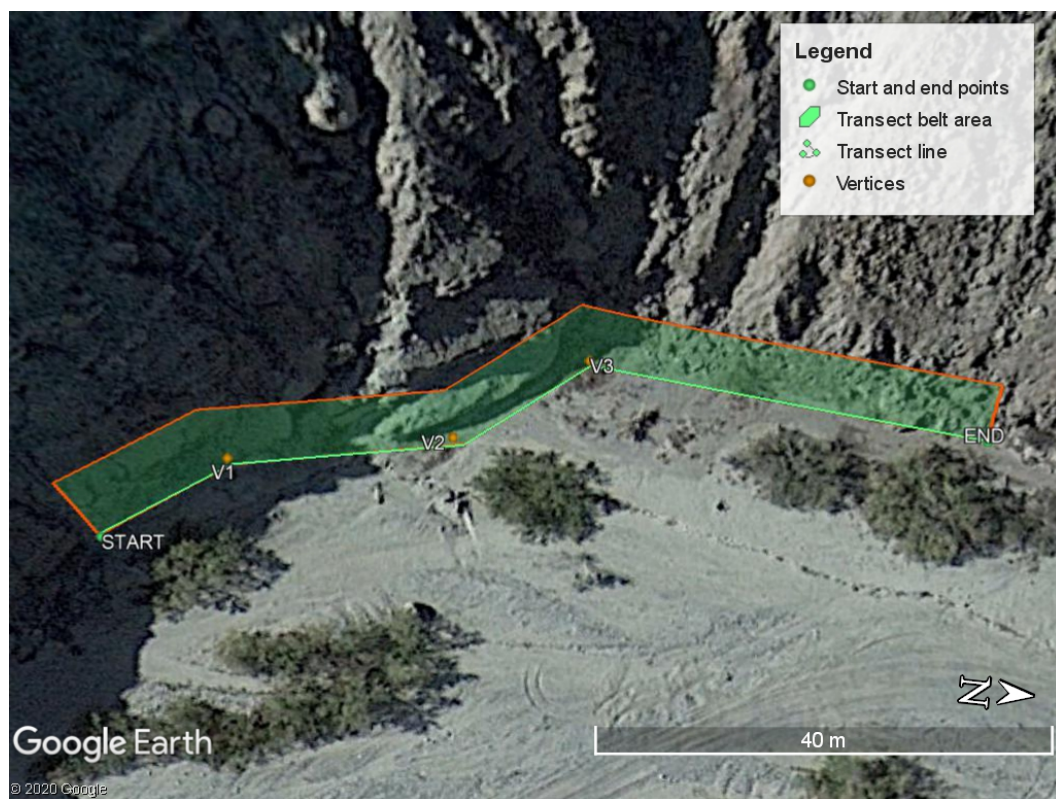


Figure 6: Diagram of vertices for *Xylorhiza cognata* survey site XYCO-04C.



Figure 7: Photo of Vertex 1 for *Xylorhiza cognata* survey site XYCO-04C.

Within the transect belt we recorded the health of each *X. cognata* individual rooted within the 5m, local disturbances, and other perennial plant species present (as with *S. greatae*). We also recorded the approximate population size of *X. cognata* outside of the belt and any other incidental populations that we encountered. Due to the terrain it was often impossible to reach individual plants within the belt, so we used the Hilti PD-42 laser range meter to determine if an individual was rooted within the belt, and binoculars to determine the health assessment when necessary.

Health scores of individual plants were recorded on the same five point scale as *S. greatae* with distinction between seedlings and adults (a seedling or juvenile being <20cm tall and no evidence of having flowered). Disturbances were also recorded within the belt transect on the same low, moderate, and high scale used in the *S. greatae* surveys. We also made note of any significant usage of land, transect details, terrain, geology, and any other unique attributes of the area. A copy of the datasheet we developed for the *X. cognata* protocol can also be found in Appendix 1.

Upon arrival at each plot we spent at least five minutes between two surveyors searching the area and adjacent canyons for individuals. If there were no *X. cognata* individuals (live or dead) present, we documented coordinates and recorded all other site data. Incidental occurrences were noted and are included in the database submitted as part of yearly monitoring activities.

Results: *Salvia greatae*

Plot Surveys

We documented 285 *Salvia greatae* individuals among 18 of the 30, 30m radius survey sites, averaging 56 individuals per hectare where the plant was present at the site. Of the 15 survey points that we repeated from 2014 vs. 2014, where it was present at five sites. The site that we did not find *S. greatae* in the resurvey, was one of the lower elevation bajada sites, where we found that 90% of shrubs to be dead, although we recorded (and established a new survey site to encompass) live *S. greatae* at a population about 150 meters north of this site.

Salvia greatae habitat was previously described as alluvial slopes, fans, and washes from 40 meters below sea level to 825 meters above. In 14 out of 30 of our plots we found that *S. greatae* was growing on steep slopes and canyon walls from 135-1011 meters in elevation. In 10 of these 14 locations the slopes and canyon walls were north, northeast facing, and the plants here occurred in dense populations, at a rate of about 72 plants per hectare. Although these individuals were smaller in size than the large individuals found in the bajadas and washes, possibly due to low moisture availability, or smaller rooting zones within the bedrock.

We found that as elevation increased the density of live *S. greatae* per site increased as well (Table 1), this is directly proportional to health, as elevation increased, the average health of the individuals in the population increased (Figure 7). As a part of the 2019 survey effort we confirmed the furthest west a population has been documented, in the Grotto slot canyons, and

the highest in elevation a population has been confirmed (1011 meters, just 106 meters shy of the Orocopa Mountains peak).

Table 1: Live *Salvia greatae* by elevation

Elevation (m)	Total Alive SAGR	Total Plots	Live SAGR /Plot
<200m	36	13	2.77
200-500m	134	13	10.31
>500m	115	4	28.75
Grand Total	285	30	

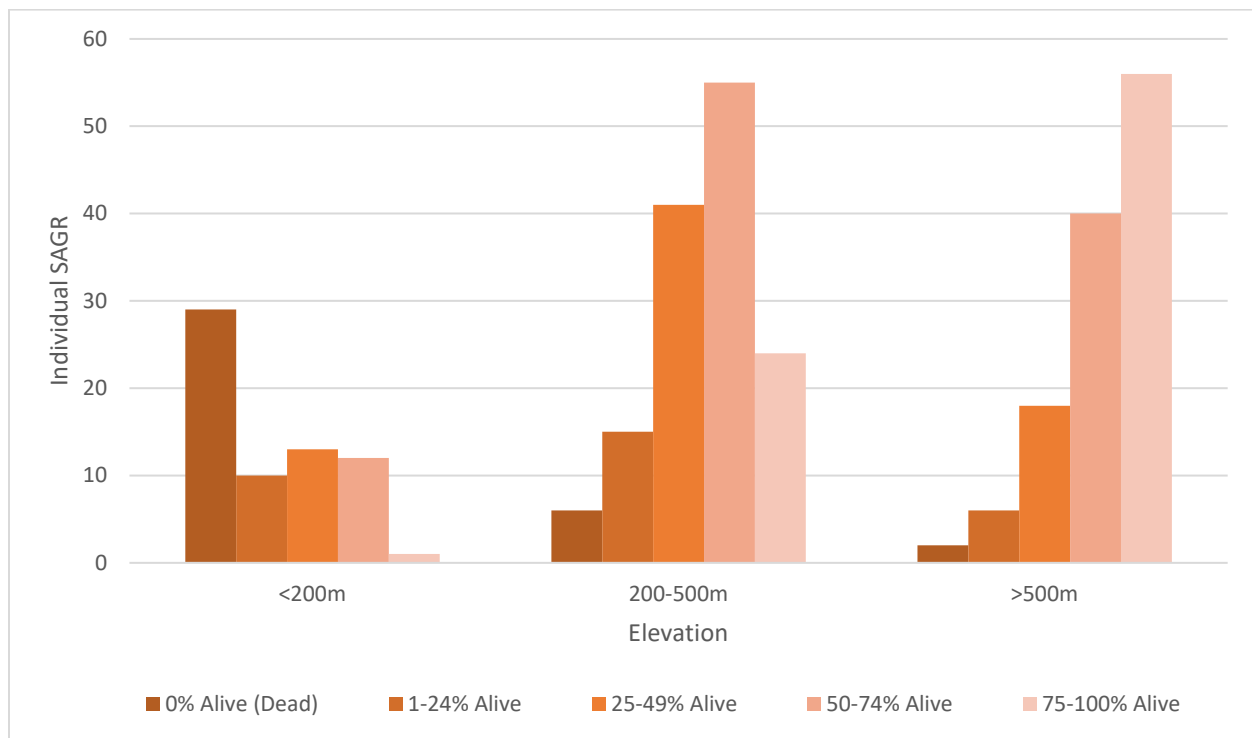


Figure 7: *Salvia greatae* health by elevation

We documented a total of 13 *S. greatae* individuals in three of the seven parallel line transects within bajadas, at a rate of about 4 individuals per hectare. Within the 3 transects, plants were not dispersed evenly across the transect, but occurred where there were undulations in topography, such as dry washes and protected outcroppings (Figure 8). Within the subset of 30m radius surveys associated with the line transects, we documented 44 *S. greatae* individuals

in five of the seven 30m radius surveys at bajada walls, averaging to about 22 individuals per hectare. These observations support our hypothesis that *S. greatae* are not distributed evenly or randomly across bajadas, rather they currently occur more densely along bajada walls, and are sparsely distributed within the braided washes throughout bajadas. This conclusion may represent a recent habitat shift; however, earlier surveys did conduct exhaustive searches for populations on the rugged upper elevation slopes within the Orocopia Mountains. Historically the largest and most easily visited population occurred in the broad alluvial fan that separates the Chocolate and Orocopia Mountain ranges at elevations of 190-160m, downslope and to the south or east of most of these surveys. As that population is contained within the Chocolate Mountains Aerial Gunnery Range, we did not measure densities or conditions of the plants remaining there. Nevertheless, higher densities (Table 1), and the distribution of individuals with higher health scores (Figure 7) occurring at elevations >200m, and especially > 500m, both indicate a current shift to higher elevations.

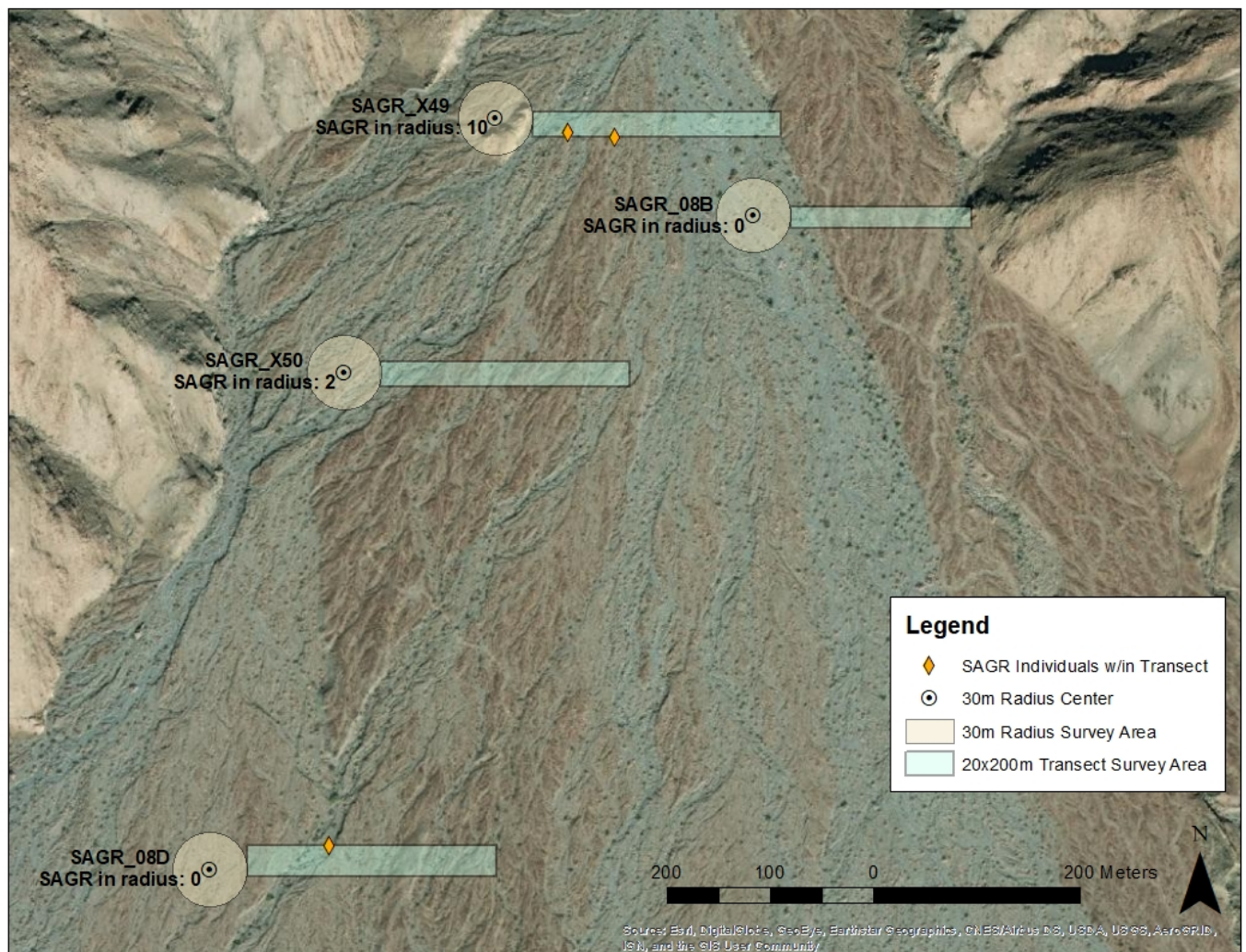


Figure 8: *Salvia greatae* radius and transect surveys of SAGR-08D, SAGR-X50, SAGR-08B, and SAGR-X49.

Extralimital Searches

We documented about 120 individuals in 11 incidental observations encountered while searching *S. greatae* habitat. Specific data for these incidental populations is included in Appendix 2.3. Eight of the incidental observations were continuations and extensions of surveyed populations, the remaining three observations were populations occurring independently. We found that nine of the 11 incidental observations occurred on steep slopes and ranged in elevation from 63-713 meters. We recorded the lowest elevation of *S. greatae* occurrence in our efforts as an incidental observation because there was only one individual present. We recommend revisiting and monitoring this individual in future surveying efforts.

Results: *Xylorhiza cognata*

We documented 723 live *X. cognata* individuals within 16 of 20 clustered survey sites, with total individuals ranging from four to 151 per site. We revisited 19 sites that were established in 2014, and found *X. cognata* at 15 of them. Our findings were consistent with the 2014 survey effort. We were able to verify all the observed populations from 2014, plus adding an additional survey site and additional incidental observations.

Indio Hills and Thousand Palms

We established one additional survey site in the western Mecca Hills and added individual surveys to two of the original sites (Figure 4). We further explored the areas around Thousand Palms (sites 19 and 20), including the oasis known as Indian Palms in the attempt to verify historic records, however, we were unable to find any *X. cognata* here.

In 2014 a single site cluster (site 16) was established at the far southeastern part of the Indio Hills, also known as the East Indio Hills Badlands. We were able to verify several of the absence points in these hills and set up an extra survey point, where we found *X. cognata* populations that were originally recorded as an incidental site in 2014.

Comparing 2014 and 2019 health and population demographic data is not as straightforward due to the differences in survey protocol and execution. When we were redesigning the protocols, we took this into account, however, in several cases it appears that individual plants that would strictly have been outside of the transect were counted, and so we are hesitant to make a direct comparison of the numbers. Despite this, several conclusions emerge about how the population demographics have changed from 2014 to 2019.

Throughout the Indio Hill sites in 2014, seedlings and juvenile *X. cognata* plants made up just over 1% of the total live population and dead individuals made up about 2.5% of the total population. In 2019 seedlings and juveniles made up just over 10% of the total live population and dead *X. cognata* plants made up 20% of the population. This change in population demographics was most likely the result of the historic drought period between 2012 and 2016,

followed by floods in 2017 and 2019; as no other explanations for the demographic shift were apparent.

Mecca Hills and Thermal Canyon

In the westernmost part of the Mecca Hills, at Thermal Canyon, we explored further up some of the canyons that fit the description of *X. cognata* habitat and established two new survey points in the slot canyons there. We were unable to access most of the points in site 13, due to construction and mining operations along the access roads (we did not incidentally observe any impact of these activities on populations), however we did investigate XYCO-13C. The GPS coordinates for this point put us in the Little San Bernardino Mountains North of I-10, and although there are historic records for these mountains and the 2014 observations indicated that there was *X. cognata* present, the photos for the point did not match the area and the habitat was not suitable for *X. cognata*, so we could only assume there was a location error. We searched adjacent canyons and ridges and found no *X. cognata*.

Mecca aster was abundant and thriving in Painted and Box Canyons (sites 1-12). We were able to repeat the 2014 surveys in these canyons. As in 2014, we were unable to locate any populations in the northern portion of Box Canyon Road, notably, as the terrain flattened out, *X. cognata* populations diminished. Our surveys reflect the accessibility of Box Canyon Road and Painted Canyon; however, based on the distributions and abundances we observed elsewhere, we are confident that *X. cognata* is abundant in the labyrinthine slot canyons that connect these two main canyons (Figure 4).

As with the Indio Hills populations, there was an increase from 2014 in seedlings and dead individuals. In 2014 seedling and juvenile *X. cognata* made up just over 12% of the total live population, while dead individuals were just over 7% of the total population. In 2019 those numbers increased to just over 18.5% for seedlings and juveniles, and almost 16% for dead *X. cognata*. Again, the most parsimonious explanation for these differences is the effect of drought preceded by flooding. No other immediate causes, anthropogenic or otherwise, were evident.

Lastly, the farthest east we have documented occurrences of *X. cognata* is Hidden Springs and The Grotto slot canyons within the Mecca Hills, where we were able to locate all of the populations that were surveyed in 2014.

There are several *X. cognata* occurrences along the Borrego Salton Sea Way (S22) in the badlands near Borrego Springs recorded in CNDDDB and CalFlora. We investigated these points and identified these plants as *Xylorhiza orcuttii*.

Results: Threats and Disturbances

As in 2014, we recorded threats and disturbances at each site for *Salvia greatae* and *Xylorhiza cognata*. We focused on OHV use, erosion (due to anthropogenic or natural means, like flooding), trampling and social trails, dumping and littering, and invasive plant species.

The remoteness and inaccessibility of both species inherently diminishes the exposure to direct anthropogenic disturbances such as OHV use, trampling, and dumping. For *S. greatae*, although populations occur near the Bradshaw Trail, we did not observe any direct impact as OHV use is restricted to the trail itself. Within the eastern Indio Hills *X. cognata* sites, we documented dumping and OHV use, and in the Mecca Hills we documented human activity (OHV use, camping, social trails, and dumping), however, observed populations were not affected as they occur primarily on the cliff walls, not in the wash bottoms where the activity was taking place.

The two most prevalent disturbances we recorded across *S. greatae* and *X. cognata* habitat was natural erosion and the presence of *Schismus barbatus*. Most populations of both species experienced natural erosion, however it is not considered a direct threat to their persistence, as both species grow in naturally erosion-prone areas. However, specific groups of *X. cognata* were likely destroyed in the 2018 flooding of Box Canyon, but the population even here is not under direct threat of major flooding events because most of the individuals occur on the hillsides and cliff faces.

S. barbatus occurred at 33 of the 73 *X. cognata* individual survey sites, and 18 of the 30 *S. greatae* sites. While we saw no direct evidence that *S. barbatus* directly competes with either plant for resources, the abundance of non-native grass adds fuel and increases the potential of wildfires.

The changes in population demographics between survey years can most likely be attributed to drought years proceeded by major flooding events. While populations of both species are not at immediate risk from either of these events, the increased ferocity of drought and flooding due to changes in climate could be a cause for increased risk to the species' survival.

Our most important findings were that both species are persisting in numbers at most of the previous sites, and recruitment is evident. However, we did find a differential amount of recruitment and mortality across elevational gradients for *S. greatae*, including a possible effect of climate change causing an upward shift in the distribution of the species. As well, we documented many *S. greatae* in areas that haven't been focal areas of past surveys, such as slopes and canyon walls, indicating additional individuals contributing to the population structure that should be considered. For *X. cognata*, natural erosion did appear to be impacting a few sites; this may be exacerbated by any increased activity such as social trails or OHV use, as well as climate change (that may increase the frequency of large, convective storms in the future), causing a negative impact to populations.

Recommendations

We redefined and separated survey protocols for *Salvia greatae* and *Xylorhiza cognata* to better capture the differences in their respective habitats. Maintaining these standardized, detailed protocols in the future will ensure that the type of in-depth periodic data required for conclusions about how each species is reacting to climate change, increased aridity, and other threats within their habitats will remain consistent.

For both *S. greatae* and *X. cognata*, sites with known occurrences of either species should be revisited with every future survey effort. In areas that the target species is absent those sites must be revisited at least one more time in the next survey before removing them from the survey effort. (CDFG, 2009). Therefore, sites that we revisited that did not have the target species in both 2014 and 2019 do not need to be revisited.

Further efforts should be made to document more high elevation populations of *S. greatae* to further understand the different habitat processes that allow populations to occur at both high and low elevations, as well as in washes and on steep slopes. To better understand the habitat conditions that allow for these plants to thrive we recommend that future survey efforts acquire permission from the Department of Defense to resurvey populations East of the Bradshaw Trail and into the Chocolate Mountain foothills, on the Chocolate Mountain Aerial Gunnery Range.

Further efforts should also be made to answer biological questions for both plants such as how long they live, age of maturation, self-pollination, primary pollinators, and biological threats to the species such as disease and insect pests.

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Appendix 1: Datasheets

[illegible]

PLOT NAME:	DATE:	TIME:	LOCATION:
CAMERA NAME:	GPS NAME:	GPS NAME:	OBSERVERS:
START PHOTO:	START UTM_E:	END UTM_E:	
END PHOTO:	START UTM_N:	END UTM_N:	START TAG #:
OTHER PHOTOS:	START PDOP:	END PDOP:	LOCATION:

START BEARING:	END CM:	TRANSECT ORIENTATION:
TRANSECT VERTICES (TRANSECT CHANGES DIRECTION ACORDING TO CANYON WASH)		
CM		
BEARING		
PHOTO		

INDIVIDUAL PLANT HEALTH ASSESSMENT (TALLY PLANTS)					
	1 (0% ALIVE)	2 (1-24% ALIVE)	3 (25-49% ALIVE)	4 (50-74% ALIVE)	5 (75-100% ALIVE)
ADULT					
JUVENILE					
TOTAL ADULT:	TOTAL JUVENILE:		TOTAL DEAD:		

DISTURBANCES (Light: <33% of stand impacted. Moderate: 33%-66% of stand is impacted. High: >66% stand is impacted)				
OHV ACTIVITY:	EROSION (NATURAL):	EROSION (UNNATURAL):	EXOTICS/INVASIVES SPECIES	L,M,H
FOOT TRAFFIC/SOCIAL TRAIL/TRAMPLING:	DUMPING/LITTER:			
OTHER (SPECIFY):				

OTHER PERENNIALS PRESENT (SPECIES)				

XYCO POPULATION SIZE (OUTSIDE OF TRANSECT)	0	1-9	10-24	25-49	50+
TALLY:					

INCIDENTAL POPULATIONS				
UTM_E	UTM_N	PDOP	# XYCO	NOTES

NOTES (ON TRANSECT DIRECTION & OTHERWISE):

Appendix 2.1: Locations and population counts for *Salvia greatae* 30m radius surveys

Survey Date	Site Name	Tag #	UTM x	UTM y	Elev. (m)	Total SAGR (Alive)	SAGR Health 1 (Dead)	SAGR Health 2	SAGR Health 3	SAGR Health 4	SAGR Health 5	SAGR Juvenile	Notes
1/23/2020	ORO_18	48	615457	3715122	786	61	1	1		17	43	0	Population and health numbers estimated, terrain too rugged to get accurate count, radius estimated. SAGR found all the way to canyon bottom, but densest on upper NE-facing slope
1/23/2020	ORO_22	96	616181	3715121	643	36	1	5	8	16	7	0	All SAGR on N facing slope, almost all of population within 30m center point, approx. 3 SAGR plants outside of radius at top of ridge. SAGR on average 1m across.
1/23/2020	ORO_28	46	617030	3712567	394	4	0	0	1	2	1	0	All SAGR on N facing slope.
1/28/2020	ORO_63	NA	611200	3717256	965	0	0	0	0	0	0	0	No SAGR present at original point or adjacent cyn (survey point), original point documented SAGR, possibly bad point or SAGR no longer present
1/30/2020	ORO_64	98	610969	3712896	407	13	0	1	5	5	2	0	All but one SAGR (on S facing slope in gully) on NNE facing slope all the way from bottom to ridge. Population continues down canyon, dry waterfall-unable to scout further.
1/30/2020	ORO_65	40	610974	3713007	388	2	0	0	0	2	0	0	Center SAGR about 2m across at widest, in wash. Second SAGR 27m SSE on edge of embankment
1/15/2020	SAGR_01A	99	623265	3710247	291	29	2	5	9	8	7	0	75% of plot in wash

1/21/2020	SAGR_01B	47	622360	3712392	400	16	2	3	4	6	3	0	About 5 SAGR up E facing slope (W side of radius) outside of circle. Circle around outcropping 3/4 way up slope, then continue up E canyon & up E side of wash. About 9 SAGR in wash & 12 in upland.
1/21/2020	SAGR_01D	44	622501	3712697	421	0	0	0	0	0	0	0	Sandstone substrate. High salt from guzzler.
1/15/2020	SAGR_03D	24	623427	3709656	275	0	0	0	0	0	0	0	No SAGR present
2/6/2020	SAGR_08B	95	613612	3709964	172	0	0	0	0	0	0	0	Mostly dead population on edge of wash. Incidental live plants dispersed NW across rocky wash, 2 live along ridge south of center point. Above population to E is LARTRI (red rock). Border is where population is dead.
10/25/2019	SAGR_08C	NA	613301	3709790	147	4	10	4	0	0	0	0	No SAGR present
2/6/2020	SAGR_08D	51	613179	3709433	105	0	0	0	0	0	0	0	No SAGR present
1/14/2020	SAGR_10A	43	612167	3709910	125	0	0	0	0	0	0	0	No SAGR present, higher out of wash. No slope, desert pavement.
1/14/2020	SAGR_10B	39	612088	3710370	168	0	0	0	0	0	0	0	Dead plants not discernable from dead SAGR, AMBDUM or PLEPLU. No SAGR outside of radius in immediate vicinity. All SAGR growing on 120 degree slope.
1/14/2020	SAGR_10C	35	612509	3710197	149	8	0	1	1	5	1	0	No SAGR present within 30m, one SAGR possibly in transect, larger population up wash, original point (612366, 3709962)
1/14/2020	SAGR_10D	NA	612370	3709962	129	0	0	0	0	0	0	0	No SAGR present
12/5/2019	SAGR_19A	NA	601499	3716722	232	0	0	0	0	0	0	0	No SAGR present
12/5/2019	SAGR_19B	NA	601291	3716329	190	0	0	0	0	0	0	0	No SAGR present
12/5/2019	SAGR_19C	NA	600837	3716172	173	0	0	0	0	0	0	0	No SAGR present
12/5/2019	SAGR_19D	NA	600587	3715983	160	0	0	0	0	0	0	0	No SAGR present

1/30/2020	SAGR_X00	0	610279	3712030	284	3	0	1	1	1	0	0	One adult SAGR on ridge 20m up on S slope. Center stake on S side of wash at base of PARFLO. Most Western extent of canyon's population extends to most Eastern point. Also end of grus/granite substrate. Sandstone starts closer to Canyon mouth.
12/10/2019	SAGR_X23	23	602176	3716005	244	10	0	0	0	2	3	5	Population continues 35m E of center point. All SAGR large, 4-5' across.
1/15/2020	SAGR_X31	31	623390	3710120	284	3	0	1	1	1	0	0	All SAGR on E facing talis slope, none in wash except one plant at ORO65. SAGR extend down & up canyon, exclusive to E slopes.
1/30/2020	SAGR_X37	37	611160	3713065	413	49	1	4	18	22	5	0	All live SAGR growing on 140degree slopes, no SAGR present in surrounding wash. Approx. 10+ SAGR outside of radius
1/14/2020	SAGR_X41	41	612391	3710092	135	12	13	1	4	7	0	0	All SAGR on hillside, none in wash
1/21/2020	SAGR_X45	45	622506	3712033	383	10	1	0	2	5	3	0	All SAGR NW side facing wash
2/6/2020	SAGR_X49	49	613403	3710040	176	10	2	4	6	0	0	0	
2/6/2020	SAGR_X50	50	613283	3709834	149	2	4	0	2	0	0	0	Center point just on S slope under trail, above AMBDUM. All SAGR on S facing slope. Population continues under S ridge, mostly in drainages.
1/28/2020	SAGR_X97	97	611354	3716946	1011	18	0	0	5	7	6	0	No SAGR present in plot, Bradshaw trail cut through half of plot, stake on NW side of willow

Appendix 2.2: Locations and population counts for *Salvia greatae* belt transect surveys

Survey Date	Site Name	Start UTM x	Start UTM y	End UTM x	End UTM y	SAGR Health	SAGR Count	SAGR UTM x	SAGR UTM y	Notes
1/15/2020	SAGR_01A	623295	3710255	623459	3710252	3	1	623305	3710260	Tossed coin for transect direction. Population continued after end.
1/15/2020	SAGR_01A	623295	3710255	623459	3710252	5	2	623301	3710250	2 plants
1/15/2020	SAGR_01A	623295	3710255	623459	3710252	3	1	623342	3710254	
1/15/2020	SAGR_01A	623295	3710255	623459	3710252	5	1	623393	3710265	
1/15/2020	SAGR_01A	623295	3710255	623459	3710252	3	1	623434	3710261	
1/15/2020	SAGR_01A	623295	3710255	623459	3710252	3	1	623444	3710257	
1/15/2020	SAGR_01A	623295	3710255	623459	3710252	3	1	623481	3710264	
2/6/2020	SAGR_08B	613638	3709962	613789	3709969	NA	NA	NA	NA	No SAGR Present
2/6/2020	SAGR_08D	613206	3709441	613412	3709446	2	1	613276	3709453	
2/6/2020	SAGR_08D	613206	3709441	613412	3709446	1	1	613276	3709453	Less than 2 years dead
1/15/2020	SAGR_X31	623418	3710116	623619	3710121	NA	NA	NA	NA	No SAGR Present
1/14/2020	SAGR_X41	612354	3710090	612153	3710085	NA	NA	NA	NA	No SAGR Present
2/6/2020	SAGR_X49	613445	3710044	613634	3710041	3	1	613462	3710029	
2/6/2020	SAGR_X49	613445	3710044	613634	3710041	5	1	613462	3710029	
2/6/2020	SAGR_X49	613445	3710044	613634	3710041	1	1	613500	3710026	Dead
2/6/2020	SAGR_X50	613310	3709834	613514	3709843	NA	NA	NA	NA	No SAGR Present

Appendix 2.3: Locations and population counts for *Salvia greatae* extralimital observations

Survey Date	UTM x	UTM y	SAGR Health	SAGR Count	Notes
2/6/2020	612793	3709159	4	1	In bajada, large
1/14/2020	612399	3710188	3	3	Toe of canyon leading to 10c
1/30/2020	609942	3711886	4	1	On rocks
1/30/2020	610495	3712095	NA	10	Irregularly spaced population on NW/W facing, non in wash bottom
1/21/2020	622640	3711437	4	15	In drainage
1/21/2020	622450	3712024	3	10	Population continues sparsley up west of point, and continues up to North side, more dense in drainages
1/21/2020	622499	3712086	4	15	SAGR plants go from the nottom of slope (not in wash) to 3/4 way up the slope
1/30/2020	610851	3713132	NA	10	East to West running canyon with stone cabin near enterance. SAGR growing at cabin on hillside and opposite canyon sides (N&S facing slopes respectively)
1/23/2020	617742	3713096	4	20	On west facing steep slope, continues but scattered further north, all on steep slopes
1/23/2020	616660	3714999	4	25	All on west facing rocky outcropping, purple rock--none on sandstone
1/23/2020	615791	3715248		10	Small clusters of SAGR scattered between ORO22 and ORO18 on ridge, almost all on N-facing slopes, some at wash bottom, 1 Or 2 on S-facing slope

Appendix 3: Locations and population counts for *Xylorhiza cognata* belt transect surveys

Survey Date	Site Name	Local	Tag #	Start UTM x	Start UTM y	Total SAGR (Alive)	SAGR Health 1 (Dead)	SAGR Health 2	SAGR Health 3	SAGR Health 4	SAGR Health 5	SAGR Juvenile	Notes
12/11/2019	XYCO_01A	Mecca Hills	33	590676	3719069	55	12	3	13	28	11	0	Last XYCO right at 100M. Population extends 20M after end transect starts at mouth of eroded slot cyn, gradually follows bottom, however cuts some curves.
12/12/2019	XYCO_01B	Mecca Hills	NA	590994	3719368	2	2	0	1	1	0	0	New point is end original transect, not worth transect
12/12/2019	XYCO_01C	Mecca Hills	NA	591246	3719410	0	0	0	0	0	0	0	No XYCO present, W facing uplift cliff
12/6/2019	XYCO_01D	Mecca Hills	NA	591075	3718999	0	0	0	0	0	0	0	Low, salt hills, next to rd., no XYCO present
12/12/2019	XYCO_02A	Mecca Hills	36	591589	3719832	14	4	0	0	6	8	0	Starts at mouth of slot canyon, can't tell last XYCO. Original point no XYCO found, bad GPS point
12/12/2019	XYCO_02B	Mecca Hills	38	591505	3719682	7	4	2	2	3	0	0	Transect travels straight up rocky gully, favoring left side of fork at top. Successive gullies to West have XYCO in them.
12/6/2019	XYCO_02C	Mecca Hills	NA	592004	3719495	5	1	0	0	3	2	0	XYO present SE of original point, cannot run transect because terrain is too difficult
12/9/2019	XYCO_02D	Mecca Hills	29	592406	3719480	100	12	6	7	10	7	70	Transect ends at end of population up canyon, 95% of juvenile XYCO had dead leaves, although some sprouting green, see photos.
12/6/2019	XYCO_03A	Mecca Hills	NA	592776	3719915	10	0	0	0	10	0	0	About 10 XYCO up rocky cliff face, avg health 4, do not continue down into wash

12/9/2019	XYCO_03B	Mecca Hills	32	593149	3720141	29	9	2	8	9	9	1	Transect starts on S side of wash at base of N facing cliff, curves E at toe until perpendicular with dying XYCO on cliff, at toe.
12/9/2019	XYCO_03C	Mecca Hills	27	593500	3720085	16	6	0	6	7	2	1	First turn point on west side of big (third) cat claw, transect starts on W side of cat claw, ends perpendicular to dead XYCO
12/6/2019	XYCO_03D	Mecca Hills	NA	592531	3719836	0	0	0	0	0	0	0	Point on rd., sandy wash with rocky hillsides, no XYCO present
12/6/2019	XYCO_04A	Mecca Hills	22	592679	3721059	56	3	1	4	10	40	1	Transect travels behind large XYCO in eroded rock pile, population somewhat continues up canyon slope, transect stays on E side of wash.
12/6/2019	XYCO_04B	Mecca Hills	30	592530	3720648	29	1	1	6	5	0	17	Transect ends in catclaw, start photo captures entire transect. Transect starts about 60m SE of original start point (not in habitat), shorts, however abundance of sprouts at base of cliff
12/12/2019	XYCO_04C	Mecca Hills	34	592792	3720211	66	3	2	5	11	38	10	Original GPS point probably bad, population center up North facing slope with erosion, healthy XYCO at bottom
12/6/2019	XYCO_04D	Mecca Hills	NA	593433	3721007	0	0	0	0	0	0	0	Steep rocky walls, no XYCO present
11/14/2019	XYCO_05A	Mecca Hills	NA	594771	3716271	0	1	0	0	0	0	0	Single dead XYCO ~5meters SW of start point, no transect run, although recommended to return to monitor
11/14/2019	XYCO_05B	Mecca Hills	14	594813	3716490	10	1	0	2	3	5	0	Transect starts at base of rocky drop off and goes S up slope with no turns
11/14/2019	XYCO_05C	Mecca Hills	20	595029	3716682	26	12	3	4	8	10	1	From start, transect travels across wash to creosote SE, population continues up canyon after transect ends
11/14/2019	XYCO_05D	Mecca Hills	21	594429	3716812	12	2	0	2	4	5	1	

11/15/2019	XYCO_06A	Mecca Hills	NA	595174	3717406	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present, historic point likely indicates adjacent canyon
11/15/2019	XYCO_06B	Mecca Hills	19	595472	3717120	10	4	0	3	5	2	0	Transect begins running S roughly along wash bottom, turns ESE following slot cyn and eventually terminating on hillside, About 5 additional XYCO uphill on N slope
11/15/2019	XYCO_06C	Mecca Hills	18	595693	3717284	9	0	0	1	5	3	0	Stake is at top of transect, starts at XYCO uphill, ends at XYCO in low area
11/15/2019	XYCO_06D	Mecca Hills	NA	596097	3717418	3	0	0	0	0	0	0	3 XYCO about 10m N of transect, not worth running line, no XYCO present at original point
11/15/2019	XYCO_07A	Mecca Hills	9	596778	3716905	5	0	0	0	1	4	0	From start point, transect travels SE at base of a NE facing cliff. XYCO outside of transect reside further up face of cliff, more XYCO after end of transect
11/15/2019	XYCO_07B	Mecca Hills	NA	597054	3716632	3	0	0	0	3	0	0	3 XYCO at bottom of cliff, populations extend into slot canyons and up rocky cliffs, about 30 XYCO total population, either original population was
11/19/2019	XYCO_07C	Mecca Hills	NA	597341	3716715	0	0	0	0	0	0	0	About 5 XYCO plants 10m up ridge, not able to sample safely.
11/19/2019	XYCO_07D	Mecca Hills	12	597194	3717084	12	1	0	0	9	3	0	XYCO continued up narrow cy, in several side cyns on all aspects. 2 juv outside of transect
11/19/2019	XYCO_07E	Mecca Hills	NA	597130	3716914	0	0	0	0	0	0	0	previously called extra 18 or h28, 10 XYCO plants in vicinity, population continues east along north facing slope

12/3/2019	XYCO_08A	Mecca Hills	16	598198	3716283	16	5	0	4	4	7	1	End of transect is previously start point, transect runs midway up slot cyn after eroded area of red rock and continues up canyon, bottom of wash curves but transect runs straight. Start tag just upslope from original point on slope, on N side of PARFLO, perpendicular to dying XYCO. Population on NW facing slope, continues up slope, favoring upper part. Road runs parallel to transect Starts at mouth of where canyon forks, goes up SW fork. HOFMIC outside of transect, ends mid wash. Start stake is behind ATRPOL and ANCFAR, runs on E side of wash, XYCO on North facing slope and starts where canyon narrows. After curve, line cuts across wash and low outcropping, ending on west facing slope. Population continues up canyon, sparsely and up N facing canyon sides
12/2/2019	XYCO_08B	Mecca Hills	6	598580	3716405	9	0	2	0	2	5	0	
12/3/2019	XYCO_08C	Mecca Hills	15	598369	3716647	10	3	1	1	5	2	1	
12/3/2019	XYCO_08D	Mecca Hills	10	598014	3716626	19	5	1	2	7	8	1	
12/3/2019	XYCO_09A	Mecca Hills	NA	599855	3717413	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present, low, rocky hillsides, some eroded steeper outcroppings, generally exposed to sun
12/3/2019	XYCO_09B	Mecca Hills	NA	599773	3717191	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present. Low, sun exposed hills.
12/3/2019	XYCO_09C	Mecca Hills	100	599746	3718229	3	0	0	0	2	1	0	Transect starts at base if PARFLO, including individual upslope of PARFLO, changes direction at XYCO at base of cliff, ending at XYCO in small cove. About 3 live plants outside, plus dead. Rocky, diverse wash.

12/3/2019	XYCO_09D	Mecca Hills	17	599072	3717136	12	0	2	2	4	4	0	Small population on east side of outcropping in large wash. XYCO on E facing slope, transect samples small population
12/11/2019	XYCO_10A	Mecca Hills	NA	600863	3720495	0	0	0	0	0	0	0	NW facing slope, low hills, no XYCO present
12/11/2019	XYCO_10B	Mecca Hills	NA	600515	3720404	0	0	0	0	0	0	0	N/NW facing slope, gneiss geology, no XYCO present
12/11/2019	XYCO_10C	Mecca Hills	NA	600293	3720099	0	0	0	0	0	0	0	Rocky exposed and eroded cliff face, E/NE facing, no XYCO present
12/11/2019	XYCO_10D	Mecca Hills	NA	600427	3719778	0	0	0	0	0	0	0	E facing slope, lower/rolling hills, no XYCO present
12/5/2019	XYCO_11A	Mecca Hills	26	600020	3715331	12	1	0	0	2	9	1	Transect runs on NW side of eroded cliff, at base of erosion and wash. Some XYCO up between eroded hill and cliff base, more scattered along edge of steep cliff base and other side of toe. 15-20 outside. Stake is on wall side of out-cropping. Transect travels SSE, cutting across narrow ridge. XYCO all along adjacent hillsides and ridges, difficult terrain, access is somewhat technical Terrain too delicate for transect Sheer cliff face, north facing, next to wash/ohv trail, likely non-habitat
12/5/2019	XYCO_11B	Mecca Hills	28	599976	3715132	17	16	1	1	7	4	4	
12/5/2019	XYCO_11C	Mecca Hills	NA	600169	3715632	6	3	0	5	1	0	0	
12/5/2019	XYCO_11D	Mecca Hills	NA	600495	3715735	0	0	0	0	0	0	0	
12/5/2019	XYCO_12A	Mecca Hills	NA	601565	3716528	11	1	0	3	6	2	0	Start stake is up cyn at base of a large, old PARFLO. Transect runs in rocky ravine, hugging the base of the curved cliff, capturing majority of XYCO on SW slope. Outside population is mostly scattered across hillside
12/5/2019	XYCO_12B	Mecca Hills	NA	601817	3716745	0	0	0	0	0	0	0	Hills surrounding cyn low, no XYCO present

12/5/2019	XYCO_12C	Mecca Hills	NA	601573	3716776	0	0	0	0	0	0	0	Steep cliff face, hills are lower, no XYCO present
12/5/2019	XYCO_12D	Mecca Hills	NA	601291	3716329	0	0	0	0	0	0	0	Point in rd., no SAGR or XYCO in walk from last point, searched cyn sides and surrounding area with no XYCO present
10/31/2019	XYCO_14A	Mecca Hills	11	585458	3724271	4	0	0	1	0	2	1	Seedling next to transect, transect travels S/SW along wash for 23m. Belt is on east side of line, small population extends up narrow cyns, some long dead >2yrs.
10/31/2019	XYCO_14B	Mecca Hills	NA	585109	3724187	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present
10/31/2019	XYCO_15A	Mecca Hills	NA	584832	3724441	0	0	0	0	0	0	0	5 min search above ridge and up wash to NE canyon, no XYCO found
10/31/2019	XYCO_15C	Mecca Hills	NA	584850	3724694	0	0	0	0	0	0	0	5 min search, no XYCO present at point.
11/4/2019	XYCO_16B	Indio Hills	NA	575651	3735281	0	0	0	0	0	0	0	5 min search, no XYCO present at point. Next to road
11/4/2019	XYCO_16D	Indio Hills	NA	576035	3734667	0	0	0	0	0	0	0	5 min search, no XYCO present at point. Heavy OHV use and dumping
11/4/2019	XYCO_16E	Indio Hills	4	575504	3735153	10	5	0	1	5	4	0	Called XYCO_X4 in notes. Transect travels uphill bearing east at base of short cliff. Transect start has one XYCO within transect
10/29/2019	XYCO_17A	Indio Hills	2,3	569454	3739041	32	13	2	8	16	5	1	DANGEROUS, do not recommend for human impact of monitoring transect, original point none found, however large population up canyons, we chose canyon closest to point. Tag at bottom

11/4/2019	XYCO_17B	Indio Hills	5	569759	3739192	13	6	0	1	1	11	0	Transect starts at base of main cyn, running until mouth of small, rocky side cyn, then turns up canyon running at base of N facing slope. 5m swath up-slope from line. Population size estimate is just for this cyn and base of main cyn included in transect start.
10/24/2019	XYCO_17C	Indio Hills	1	569349	3739378	46	5	0	5	16	16	9	Transect starts perpendicular to top most XYO of SW fork, roughly follows center of bottom of cyn. At vertex 3, go up right cyn, ended at last XYCO in transect
10/29/2019	XYCO_17D	Indio Hills	NA	569263	3739182	0	0	0	0	0	0	0	5 min search, no XYCO present at point.
11/5/2019	XYCO_18A	Indio Hills	13	568370	3740622	4	0	0	0	2	2	0	Population continues beyond end, however cyn in impassable
11/5/2019	XYCO_18B	Indio Hills	NA	568529	3740303	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present, went 50m up canyon
11/5/2019	XYCO_18C	Indio Hills	NA	568708	3739882	1	0	0	0	1	0	0	Single XYCO up adjacent South canyon along with long dead shrubs
11/5/2019	XYCO_18D	Indio Hills	NA	568376	3739742	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present
11/6/2019	XYCO_19A	Thousand Palms	NA	561604	3745957	0	0	0	0	0	0	0	Explored SE up cyn, no XYCO present
11/6/2019	XYCO_19B	Thousand Palms	NA	561911	3746133	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present
11/6/2019	XYCO_19C	Thousand Palms	NA	562229	3746021	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present
11/6/2020	XYCO_19D	Thousand Palms	NA	562512	3745793	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present
11/5/2019	XYCO_20A	Thousand Palms	NA	560574	3744552	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present

11/5/2019	XYCO_20B	Thousand Palms	NA	562495	3744322	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present
11/5/2019	XYCO_20C	Thousand Palms	NA	560262	3744045	0	0	0	0	0	0	0	5 min search up wash and adjacent canyons, no XYCO present
10/31/2019	XYCO_21A	Mecca Hills West	7	586787	3724822	14	1	2	1	6	5	0	Called XYCO_X7 in notes.
10/31/2019	XYCO_21B	Mecca Hills West	8	586476	3724845	8	0	0	0	2	6	0	Called XYCO_X8 in notes. Transect starts in cyn E of main cyn and continues to second bend mid cyn. Travels S/SW, population continues down wash.

Appendix VIII –
2020 Monitoring Results for the
Triple Ribbed Milkvetch
(*Astragalus tricarinatus*) within the
Coachella Valley MSHCP Area

Coachella Valley Multiple Species Habitat Conservation Plan
& Natural Community Conservation Plan

2020 Monitoring Results for the Triple Ribbed Milkvetch (*Astragalus tricarlinatus*) within the Coachella Valley MSHCP Area



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TRIPLE RIBBED MILK VETCH MONITORING

Triple-ribbed milkvetch, *Astragalus tricarlinatus* A. Gray (Fabaceae) is a short-lived perennial herb endemic to southern California, occurring along the ecotone of the Mojave and Colorado Deserts in the San Bernardino and Little San Bernardino Mountains, although there is a disjunct occurrence in the Santa Rosa Mountains (USFWS 2009; Fraga and Pilapil 2012; Jepson Flora Project 2017). It has also been reported from further east in the Orocopia Mountains by Barneby (1959, 1964), but this is unverified (USFWS 2009; Bell et al. 2017). A specimen collected by M. F. Spencer dated April 6, 1921 indicates the locality as “Chuckwalla Mtns” [sic] and this resides at Harvard University Herbarium where it is indexed and an image is available online (Harvard University Herbaria and Libraries 2020).

In 1998, triple-ribbed milkvetch was listed as endangered by the United States Fish and Wildlife Service based in part on the state of knowledge about the species at the time-- that it occurred as small, ephemeral occurrences on benches along desert washes and canyon bottoms; such occurrences are now believed to be waif or deme populations (Barneby 1959; Sanders 1999; USFWS 2009; Fraga et al. 2015). Core habitat is now recognized as further upland in topographically-rugged, friable soils, often in upper watersheds, and so difficult to reach (White 2004; USFWS 2009; Fraga et al. 2015; Bell et al. 2017).

We initiated study of this species as part of the monitoring of protected species, including triple-ribbed milkvetch, under the Coachella Valley Multiple Species Habitat Conservation Plan (Coachella Valley MSCHP Permittees), with the aim of collecting data that will contribute to the long-term persistence of self-sustaining populations (Coachella Valley Conservation Commission, 2016). Our broad objective is to evaluate threats to persistence of the known occurrences of this species in the San Bernardino and Santa Rosa Mountains, within the CVMSHCP. Such threats may include human disturbance, invasive species, natural stochastic events, and climate change. Past research by the University of California, Riverside Center for Conservation Biology (CCB) found that invasive plants may reduce flowering and seed set in this species (Heintz *et al.* 2018). Further knowledge about the degree to which such threats impact triple-ribbed milkvetch can lead to appropriate land management protocols and an update of the listing status of this species (Amsberry and Meinke 2007; Fraga and Pilapil 2012; Fraga *et al.* 2015).

This study also aims to contribute information to the USFWS Special Status Assessment of this species, a recovery plan (none has been produced to our knowledge) and as a follow up to a genetic analysis done by Fraga and others (2015). New genetic information and analysis may help determine a) population genetic variation and viability of populations and b) to determine what function, if any, that the waif (bottomland) groups serve in terms of their contribution to local and regional gene flow. Is the Santa Rosa group of plants in decline and of low genetic variability? We seek to find out what the genetic structure is between these distinct, isolated, small populations, and what is their relatedness to the waif populations. Are the waifs functioning as “genetic bridges” or are they simply a genetic dead end, not contributing further to sustained, permanent source populations? As well, sampling and determining the presence and type of root symbionts (rhizobia, nitrogen-fixing bacteria), and contrasting these among upland and waif populations may help elucidate the factors causing fluctuations in the populations of waifs, as suggested by Amsberry and Meinke (2007). This information will aid in the determination to what extent waifs are necessary for population viability, and further, if threats to waif populations represent in fact any threat to the recovery of the species.

The University of California, Riverside Center for Conservation Biology (CCB), carried out surveys for triple-ribbed milkvetch in order to meet monitoring and management goals within the CVMSHCP

(Coachella Valley MSCHP permittees). This is the fourth year of monitoring and the second year of permitted sampling. In the 2019 spring season (an average-to-above average rainfall year), most areas that had supported plants in the past continued to support living individuals, and the plants found on the whole appeared healthy. It should be noted that many occurrences support just one plant, and there were some recent locations that no longer appeared to be supporting plants (data provided to the California Natural Diversity Database, CNDDDB). Many seedlings were noted at several sites with larger numbers of plants. Despite searches at wash and bottomland locations known to support plants in the past, and perhaps due to the heavy rainfall events of October 2018 and February 2019 and apparent scouring of wash bottoms, we did not locate any of these “waif” plants separated from upland groups. We sampled individuals from seven local catchment areas, for a total of 39 individuals in the first study year. Subsequently along with study partners, we obtained samples from three more areas (Little Morongo Canyon, Eureka Peak area, and the Covington Flats Area).

Several areas of interest were not accessed in 2019 due to the limits of time, and were prioritized for revisits this year, spring 2020. Additionally, we received funding for the genomic analysis of samples and determination of root symbionts (rhizobia) from a USFWS - Joshua Tree National Park Interagency Agreement/California Ecosystem Studies Unit agreement.

Objectives

Surveys for triple-ribbed milkvetch were carried out in order to monitor the status of triple-ribbed milkvetch populations within the CVMSHCP and to provide recommendations for any necessary management. The outcome of this multi-year project is expected to identify whether populations are genetically isolated, whether various populations appear to be stable and viable, and how the waif populations are related to upland populations. This information will help in recovery of the species by identifying whether any significant threats to waif occurrences pose any danger to the recovery of the species, an important focus, identified by Fraga and others (2015). It may be that the waif individuals are not self-sustaining populations, they do not contribute to population structuring spatially or temporally and so identified threats (per USFWS 2009) to the species pose no danger to the sustainability of populations as a whole. Or, these waif individuals may be a key linkage between isolated upland populations. Information about the isolated upland and bottomland Santa Rosa occurrences, in particular, will help guide future monitoring and management of that population. If this population has genetic variation comparable to other populations and apparent population sizes appearing to be steady since other surveys by Bell and others (2017), there may be reason to be less concerned about the probability of persistence of this population.

This season, we increased our effort to search for and sample the bottomland populations, to discover if any plants had established after the major rainfall events of October 2018 and February 2019. Thus, the specific objectives of this second year of sampling were to: increase the number of individuals sampled across space, search for and sample bottomland plants, sample whole seedlings for identification of rhizobia across soil types (as permitted), sample soils for analysis, and collect seeds from a limited number of areas, for further experimentation with rhizobial symbionts, in partnership with the lab of Dr. Joel Sachs and his post-doctoral scholar, Dr. Lorena Torres-Martinez at UCR.

Methods

Background

In the fall of 2018, we applied for a USFWS Recovery Permit to permit sampling in support of our conservation research on triple-ribbed milkvetch, as well as permission to carry out the research within designated BLM and USFS Wilderness. The Recovery Permit and Letters of Authorization were received in spring, 2019. The BLM/USFS Letter was renewed for the 2020 season. We applied and received a modified Recovery Permit in spring 2020 to allow for increased flexibility in sampling leaf tissue (to allow for sampling of solo individuals at some sites), seed collection, and to add personnel to the Recovery Permit. Permission was granted by the University of California, Riverside College of Natural and Agricultural Sciences administration, the local site managers for The Wildlands Conservancy and the Friends of Big Morongo Canyon to allow ongoing research during the COVID-19 pandemic.

Survey Area

Surveys were conducted during February to July 2020 within the Upper Mission Creek/Big Morongo Canyon Conservation Area (UMCBMC) and the Santa Rosa and San Jacinto Mountains Conservation Area (SRSJM), as well as just to the north of the CVMSCHP area, aimed at more complete coverage of the occupied habitat, especially some of the earliest-documented sites that were relevant to the ESA listing (Fig 1). Due to restrictions surrounding the COVID-19 pandemic, several sites without research exemptions for access and/or requiring lengthy hikes such as Joshua Tree NP partner sites and the Wathier Landing area were not accessed.

Surveys were performed by two experienced individuals generally traversing wash habitat enroute to upland populations, using binoculars and focusing on areas exhibiting the typical “distressed granite” substrate in which triple-ribbed milkvetch appears to thrive (White 2004). Surveys included sites of previous surveys by UCR CCB, herbarium specimen localities, localities from the CNDDB, and locations provided by local experts and the Rancho Santa Ana Botanic Garden.

Triple-ribbed milkvetch study area within the Coachella Valley, California

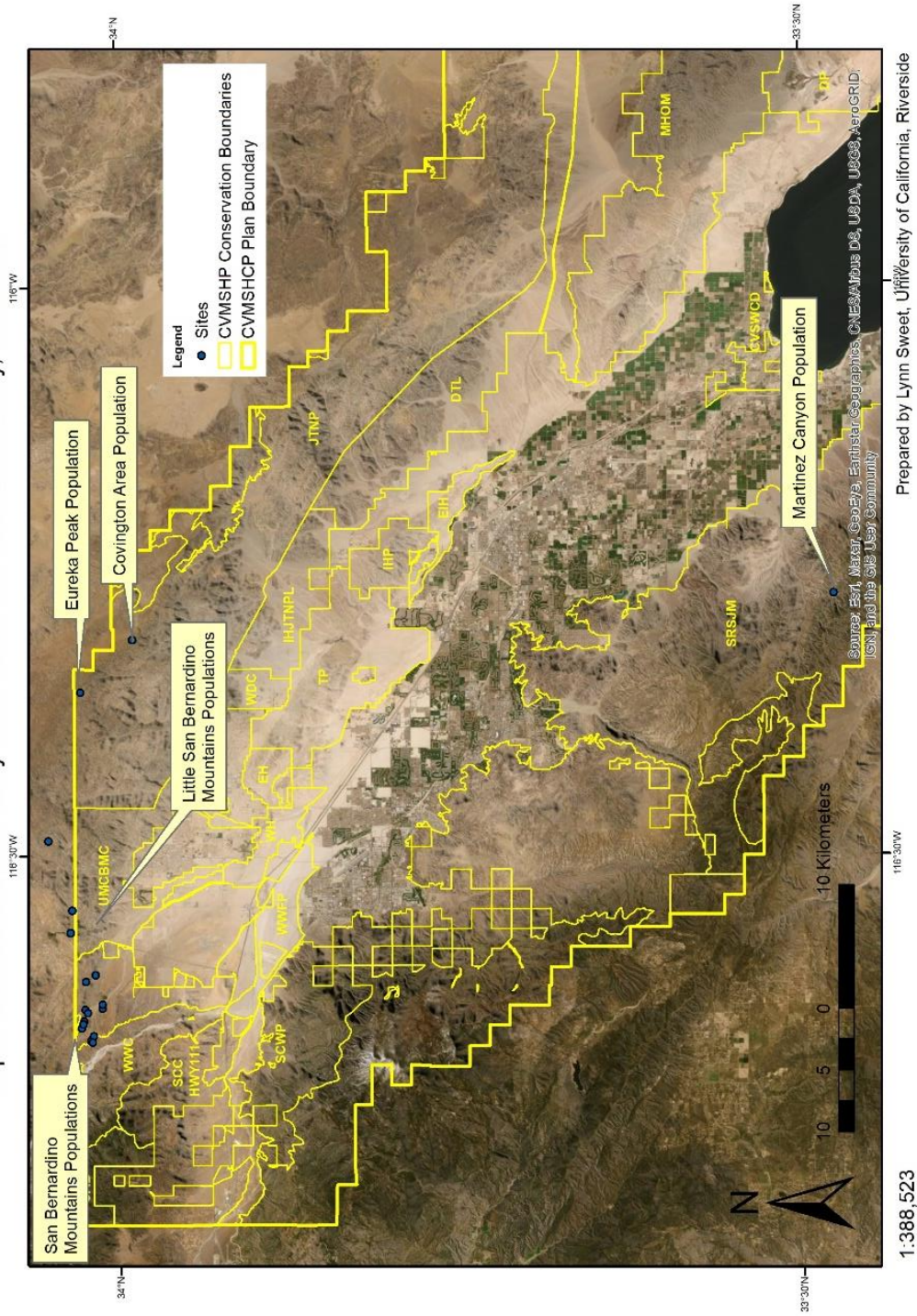


Figure 1: Triple-ribbed milkvetch populations visited during 2019 and 2020 within the known range of the species. Also shown are surveys by study partners that may contribute information to the range-wide genetic

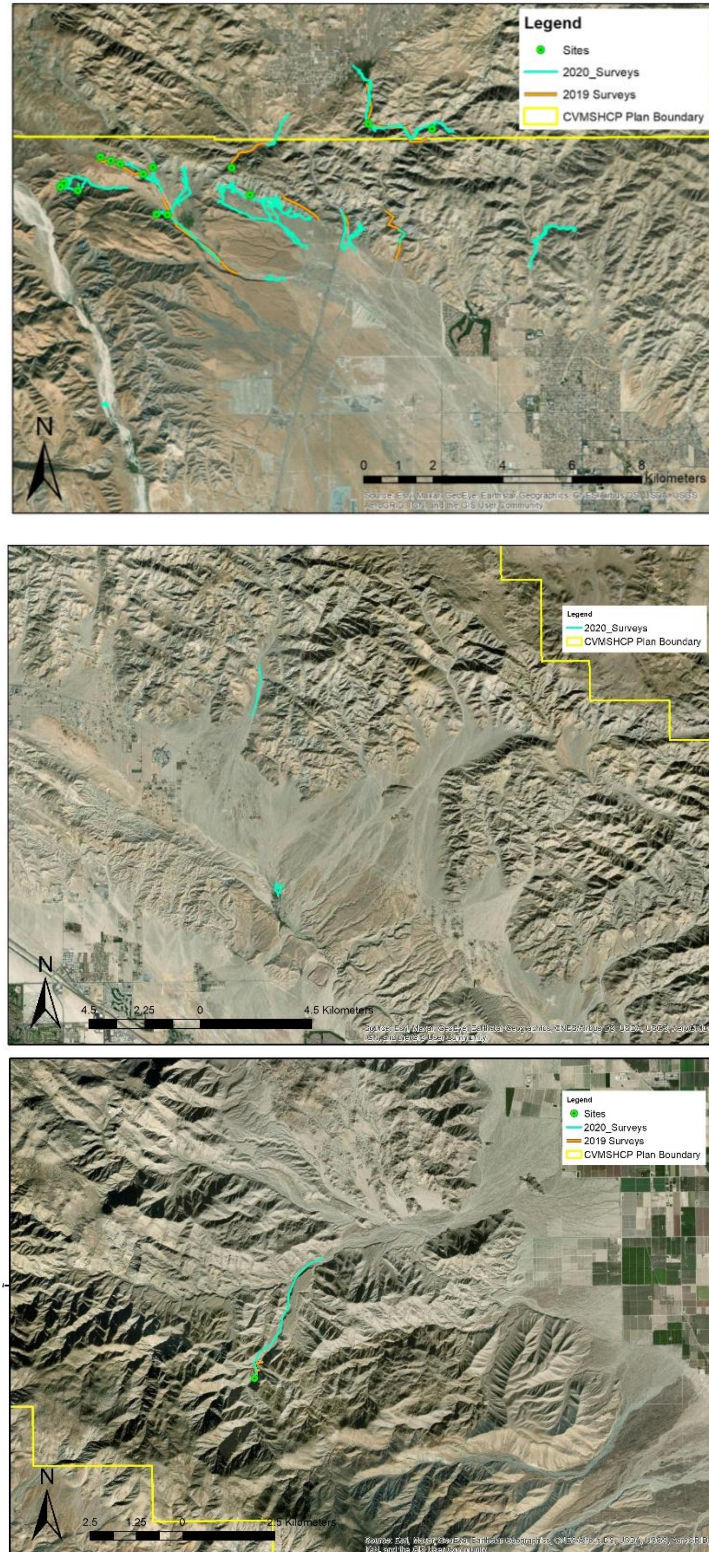


Figure 2: Triple-ribbed milkvetch surveys and populations visited during 2019 (orange) and 2020 (light blue) within the northwest range of the species, Upper Mission Canyon and Big Morongo Canyon (top), Thousand Palms Canyon and East Deception Canyon (middle) and within the Santa Rosa and San Jacinto National Monument Conservation Area (bottom). All survey lines are approximate.

Data Collection

Between February and July of 2020 (including revisits to sample seeds), we located the plants for study, recorded abundance, and sampled tissue for analysis and collected seeds for experimentation. When surveying a population every attempt was made not to disturb the area more than was necessary, in accordance with the details of our Permit and Letter of Authorization, referencing Center for Plant Conservation guidelines (CPC 2017). For each initial survey, GPS tracks were generally recorded to document the survey area (shown in Fig 2). We documented all plants, in known and previously unknown locations as GPS point locations (WGS84 UTM 11S) recorded using Avenza PDF maps (www.avenza.com) on an iPhone 8, with no additional receiver. In many cases, the GPS signal was poor due to multipath error and limited sky visibility in canyon locations, and an effort was made to correct the point locations post-collection using georeferenced aerial photos. We estimated the total number of individuals present per grouping and recorded this as a range based on the density of plants in the area accessed and the estimated extent of the grouping, except where a single number is reported, indicating that individuals were counted. We did not survey the full extent of the grouping at every location, nor revisit all study plants from past surveys due to the survey focus on plant sampling. Where necessary, coordinates were projected/estimated from a safe location, as noted. Locations listed also include absences and all of these were submitted to the CNDDDB.

Plant sampling was performed by permitted individuals, following the guidelines for the number of samples and the method of collection. We took a photo of each study plant, noted the life stage and GPS coordinates. We sampled leaf tissue from mature plants or plants with >10 leaves and/or sampled a whole seedling, including root tissue, and additionally collected seeds. Leaf samples were placed into small centrifuge tubes kept fresh within a vacuum-insulated canister with (water) ice. Samples were transferred to a standard freezer (-18°C) and then transported on dry ice to a -80°C freezer in the lab of research partners Professor Joel Sachs and Dr. Lorena Torres-Martinez at UCR. When sampling whole seedlings for leaf tissue and roots, water was applied to the plant while a small hole was excavated around the root, until nodules were seen, or the taproot was able to be removed (Fig 3). We placed the seedling in wet paper towels and transported them unfrozen but below ambient temperature in a soft cooler, stored in a refrigerator and transported within 1 day to Dr. Torres-Martinez for isolation of the symbionts and long-term storage of the leaf tissue.

Rhizobia were isolated from the nodules of each young seedling and clonal cultures were grown and archived for whole-genome sequencing to identify species of the symbionts. Nodules were surface sterilized with bleach and rinsed with sterile water before being crushed with glass rods and the contents were plated on glucose-based rhizobium-defined medium (RDM) with cyclohexamide as an antifungal and bromothymol blue as a pH indicator. Among the resultant colonies, we selected those that grew on arabinose gluconate medium (MAG) and GRDM but not on Luria-Bertani (LB) medium within 5-8 days after inoculation. Selected colonies were archived in a 1:1 solution of liquid MAG and glycerol at -80C for further studies.

We collected seeds from plants later in the season, May-July on revisits to our sampling locations when pods were dry and splitting. Plants with both tissue and seeds collected at different times were matched using GPS points and visually using substrate/woody debris landmarks in photos of the plants. Seeds were generally collected into small envelopes, and we extracted seeds from the fruits in lab within 2 months, and transported them to Dr. Torres-Martinez.

One set of samples was processed in 2019 as a test of the methodology. Plants were genotyped using genome-wide molecular markers through reduced representation sequencing (RR-Seq). Briefly, this is a genotyping-by-sequencing (GBS) technique where the DNA was digested with the restriction enzyme ApeKI and resulting fragments were sequenced in an Illumina-4000. Prior to library preparation for the RR-seq, genomic DNA was extracted from the fresh leaf tissue using a Qiagen kit. DNA quantity was assessed with a Qubit fluorometer. Library preparation and sequencing were performed at UC Davis Genomics Core.

Results

All of the upland populations that we revisited from previous surveys supported plants in spring 2020. We located/confirmed 24 areas supporting groups/populations of plants during the survey period, and presumed absences at 13 others (Table 1). Of the presences, nine areas were either new occurrences or were adjacent and not within previous CNDDDB “Element Occurrence” (EO) polygons. This year we located “wash bottom” plants in an area of the North Fork of Mission Creek nearby upland populations (Fig. 4), an area where plants were not detected by our team in 2016-2019. Surveys in search of plants at and around older localities within the warmer edge of the range of the plant (canyon mouths and foothills in Desert Hot Springs and the Devils Garden foothills) resulted in just two plants found. In the Indio Hills, which we had not previously surveyed, we visited the site of the 2019 Thousand Palms Canyon specimen identified by Center for Natural Lands Management (CNLM) staff and confirmed the presence of a recently-dead plant. Searches in the East Deception Canyon did not result in any findings. Surveys within the Mecca Hills and Orocopia Mountains for Mecca aster and Orocopia sage, by searchers trained to identify *A. tricarinatus*, also did not result in any incidental sightings of this species within those areas (see report on Mecca aster and Orocopia sage for the description of the areas visited).

We gathered 81 leaf tissue samples for analysis, from populations spanning the range of the species, bringing the two-year total to 134 samples. Seeds were gathered from seventeen live plants paired with leaf samples, and four dead plants for which we were unable to collect leaf tissue. Unfortunately, the Thousand Palms Canyon plant did not have any seeds to collect. Whole seedlings with roots were collected from five different areas.

We successfully isolated a total of 60 rhizobial isolates, that based on colony coloration and growth in GRDM media are from the *Mesorhizobium* genus, as it has been found in other *Astragalus* species worldwide (Yan *et al.* 2016). Further molecular work will inform the species identity, and functional diversity associated with the soil type where they were collected.

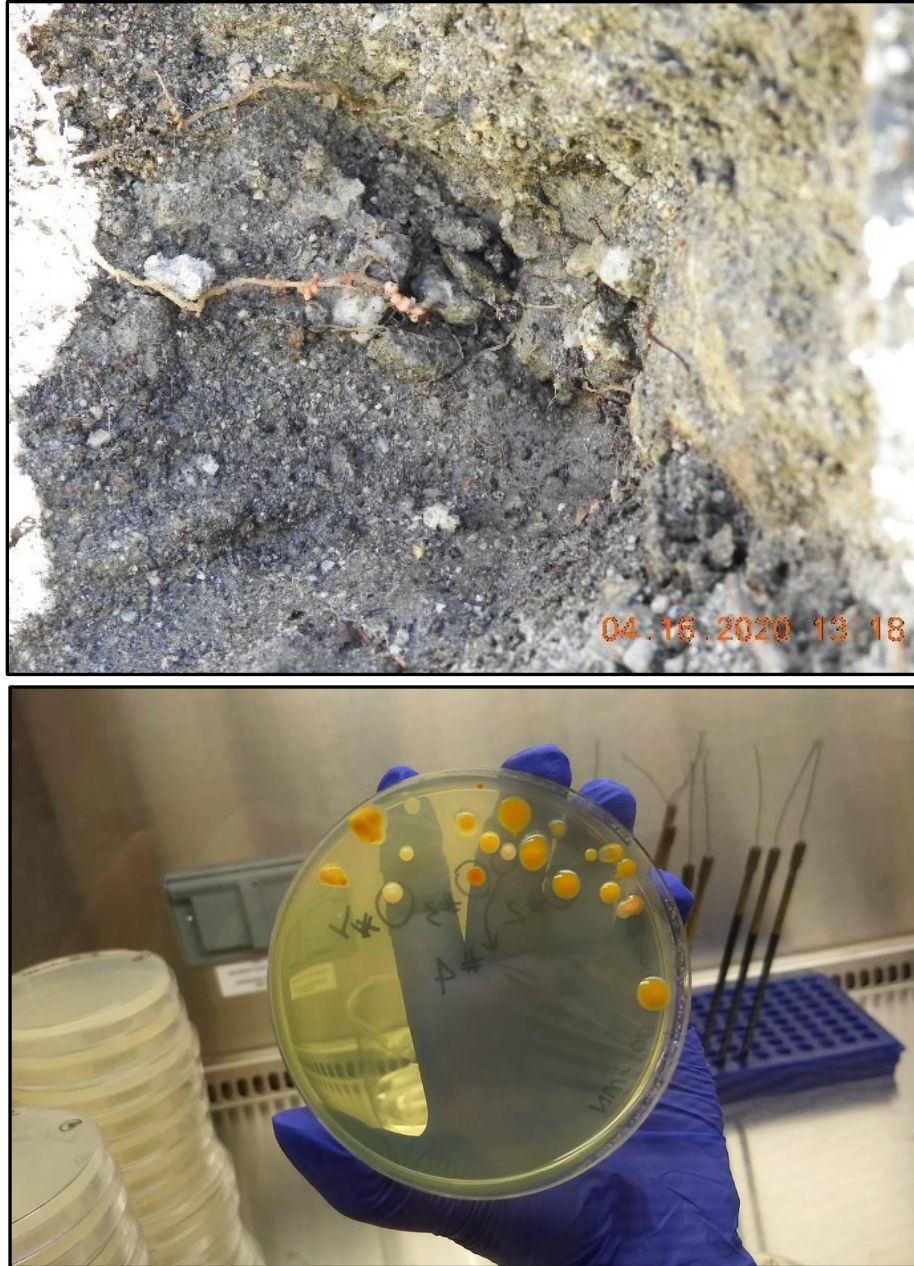


Figure 3: Upper photo: *Astragalus tricarinatus* root nodules indicative of the development of rhizobial symbiont colonies for extraction of nitrogen from the soil (Photo, Larry Heronema). Lower Photo: Colony coloration and shape in GRDM of rhizobia isolated from a single nodule of *A. tricarinatus* from a seedling collected in the wash at the North Fork of Mission Creek. The diversity of colonies from one nodule suggests the association of *A. tricarinatus* with a broad-range of Mesorhizobium symbionts (Photo, Lorena Torres-Martinez).

Table 1: Locations surveyed for *Astragalus tricarlinatus* during the study period, and estimated number of plants seen at each location, including absences. Note that UTM coordinates for surveys indicate a general location, and all coordinates are truncated (=10,000 m accuracy); precise coordinates were provided to agencies as appropriate. EO's refer to any prior Element Occurrence number assigned for the CNDDDB.

Area	Site	Type	CNDDDB EO	WGS84 11S UTM E	WGS84 11S UTM N	# Milkvetch Seen or Est.	Samples
Big Morongo Canyon	Big Morongo Canyon	Location	77302	539800	3766400	~2	Y
	Big Morongo Canyon	Location	77302	539800	3766300	3	
	Big Morongo Canyon	Survey	117318	542200	3766100	0	
	Big Morongo Canyon	Survey	18928	539900	3767300	0	
	Big Morongo Canyon	Location	77302	539900	3766200	20-30	Y
	Big Morongo Canyon	Location	Near 77302	540100	3766100	1	
	Big Morongo Canyon	Location	77302	539800	3766300	1	Y
	Big Morongo Canyon, County Line	Location	New	541700	3766100	5	Y
	Big Morongo Canyon, County Line	Location	New	541700	3766100	2	Y
	Big Morongo Canyon, County Line	Location	New	541600	3766100	1	
	Big Morongo Canyon, south end	Survey	NA	540900	3763000	0	
Dry Morongo Wash	Dry Morongo Wash, north	Survey	NA	537000	3765700	0	
	Dry Morongo Wash, side canyon	Survey	NA	539500	3763100	0	
	Dry Morongo Wash, south end	Survey	18927	539400	3762800	0	
East Deception Canyon	East Deception Canyon	Survey	80591	562900	3753900	0	
Indio Hills	Thousand Palms	Location	77298	563600	3745500	1	
Little Morongo Canyon	Little Morongo Canyon	Survey	117317	545600	3763200	0	
Mission Creek Canyon	Devils Garden Foothills	Location	New	537000	3763900	1	Y
	Devils Garden Foothills	Location	New	535900	3764500	1	Y
	Mission Creek (Wash at North Fork)	Location	between EO 87504/32960	533500	3764700	50-75	Y
	Mission Creek Canyon	Survey	60718	534400	3763300	0	
	Mission Creek Canyon	Survey	60718	534300	3763700	0	
	Mission Creek Canyon (Stone House Adjacent)	Location	60718	534100	3763600	1	Y
	Mission Creek Canyon (Stone House Adjacent)	Location	60718	534100	3763600	3	Y
	Mission Creek Canyon (Stone House)	Location	60718	533800	3763600	9	Y
	Mission Creek Canyon Northeast	Survey	NA	534600	3765000	0	
	Mission Creek, North Fork	Location	87504	532800	3765100	10-20	Y
	Mission Creek Pass	Location	Near 87534	531100	3764700	1	Y
	Mission Creek Pass	Location	Near 87534	531100	3764500	4	Y
	Mission Creek, True North Fork	Location	32960	532700	3765100	1	
	Mission Creek, True North Fork	Location	32960	532700	3765100	1	
	Mission Creek, True North Fork Spire	Location	87534	531500	3764300	20-30	Y
Santa Rosa Mountains	Santa Rosa Mtns/Martinez Cyn	Location	117314	567600	3704300	28+	Y
	Santa Rosa Mtns/Martinez Cyn	Survey	117312	568400	3706100	0	
Whitewater Canyon	Mission Creek Pass, Whitewater side	Location	87534	531000	3764400	5+	Y
	Whitewater River channel	Survey	53494	532300	3758200	0	

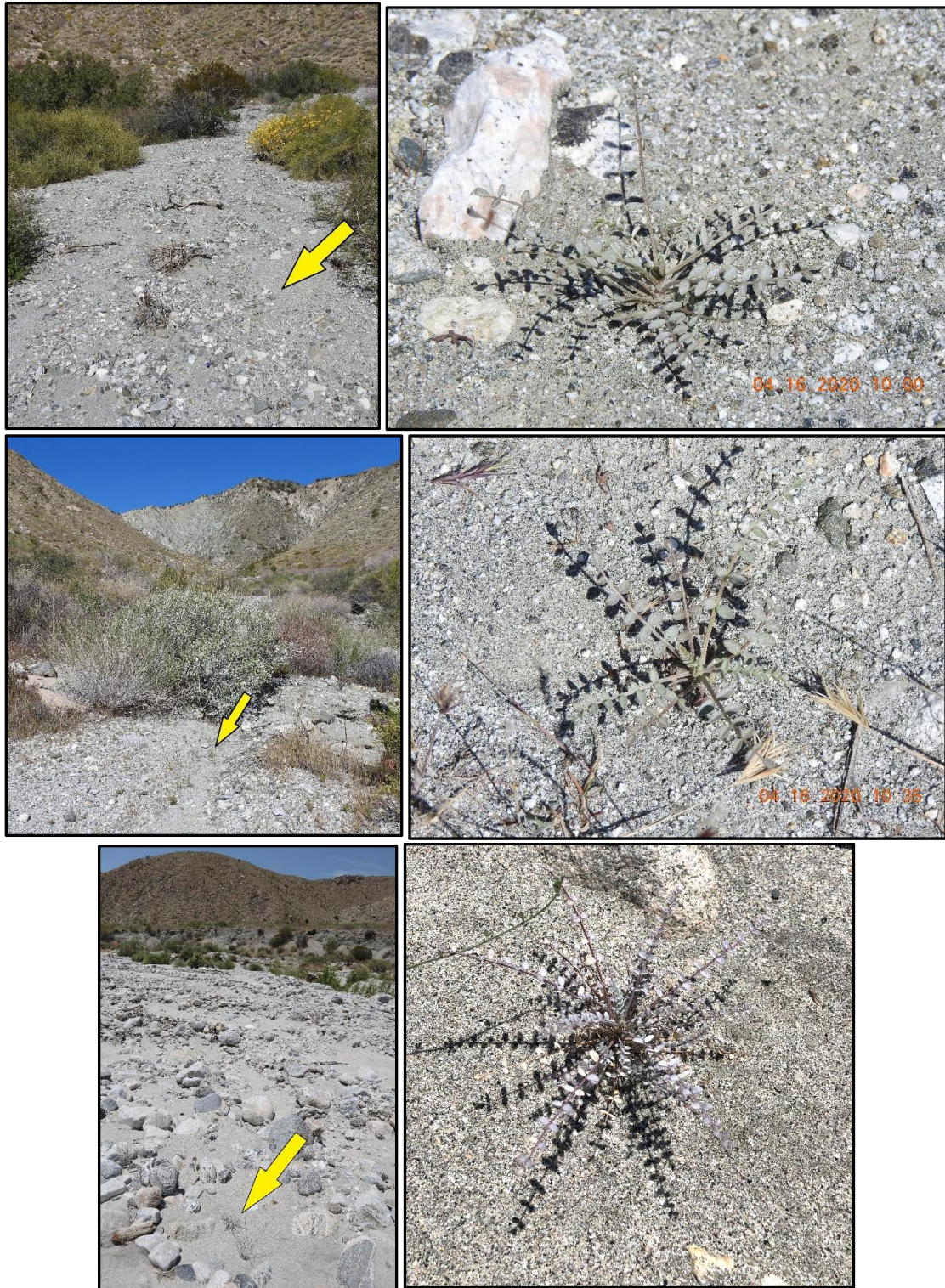


Figure 4: Paired landscape and close-up photos of *Astragalus tricarinatus* seedlings in wash bottoms along a Mission Creek North Fork minor wash (first, second pair) and main wash (last pair).

Although we did not strictly revisit the study plants from the 2017-2018 study, many of the plants from prior years were observed to be still live or had died but were still evident on the landscape, and new plant establishment was clearly evident. We detected apparent first year seedlings on the landscape starting in March, through April, and noted some persisting into May as small plants (Figure 4, bottom photos), showing pigmentation changes due to extreme radiation or prior to senescence, but lacking inflorescences. We saw what appeared to be a class of second year plants, some very small plants that were flowering that lacked old rachises, but it cannot be determined definitively if they recruited this year or prior. We did not follow plants from seedlings stage through the season, but also did not observe any apparent first year seedling flowering at the end of the season. Plants were seen flowering from the beginning of March until at least late April, when searches concluded. We collected seeds from live plants starting in late March (early), several times in April (peak), once in mid-May (past peak), and in July (well past peak). We noted that plants visited in early July in Big Morongo Canyon had fully senesced, likely the predominant pattern, but all populations were not visited during the summer.

A few, very limited, potentially negative biotic interactions were noted. We did not reinvestigate last year's observation of herbivory by aphids and *Largus* bugs. At the Big Morongo site, three plants were present that had brown patterning affecting their pods and leaves (Fig. 5). These plants were located very low on a rock outcrop, adjacent to thick stand of invasive grass and shrubs, about 20m from the creek bed, an unusually humid microsite for this plant. Samples were approved by USFWS to be given to a plant pathologist at UCR, but due to COVID-19-related lab restrictions, definitive results have not been reached, and it has not been confirmed that the origin of the issue was a pathogen. We can conclude that whatever was causing the patterning on the pods did not prevent seed formation. In terms of other impacts, we did not note any adverse effects of herbivory on leaves, however, at the upland North Fork of Mission Creek site, we noted that one single plant showed unusual amount of herbivory on the pods, which may have affected seed formation, but did not entirely prevent it. It should be noted that this plant was somewhat buried by the excavation of a fossorial mammal burrow.



Figure 5: *Astragalus tricarinatus* pods showing the unusual brown patterning found on plants in Big Morongo Canyon (left), and a high amount of herbivory (right).

In terms of natural symbionts, we did not conclusively identify pollinators and seed dispersers, although our staff entomologist, Scott Heacox briefly observed a native bee seen presumably pollinating a plant at the Mission Creek site (Stone House Adjacent), and Lynn Sweet observed a European honeybee (*Apis mellifera*) on a flower at the Big Morongo Site (Fig. 6). As in past years, canyon wrens, rock wrens and lesser goldfinches were commonly seen within the habitat in proximity to the plants. On examination, the

Pods have a triple hook structure when incompletely dehiscent as they tend to fall, and pods were observed caught on microfeatures of the habitat (soil mounds, roots, branches). As mentioned, Dr. Torres-Martinez was able to isolate rhizobial symbionts from root nodules, and they will be further analyzed.



Figure 6: A European honeybee (*Apis mellifera*) on the flower of *Astragalus tricarinatus* in Big Morongo Canyon.

Within the habitat, there was clear evidence of significant ephemeral streamflow associated with probably the October 2018 and February 2019 major precipitation events and possibly subsequent events, including washouts within canyons, scouring, new runnels and gullies. Particularly in the steepest upland habitat, erosional processes were evident, and unvegetated substrate was exposed. Many canyon walls looked recently-eroded and debris was piled at the base of slot canyons, although this is likely a common process of the habitats in which it occurs. *A. tricarinatus* plants were found recruited in the North Fork of Mission Creek on newly-eroded material within the dry creekbed, as well as on canyon walls that were clearly eroding. At this site as well, plants that had been seen on small canyon walls in previous years had disappeared, and the slope was barren. Information from the CNLM staff at Thousand Palms Preserve and our assessment of the location/phenology of that specimen indicates these plants possibly established as waif seedlings or seeds from the October floods of 2018, washed down from East Deception Canyon, that sprouted and grew to a small size in the spring of 2019, flowered and set a limited amount of fruit, senesced in the summer of 2019, and then regrew only a limited amount of foliage in spring 2020 and died.

In terms of threats to the species, we noted OHV tracks on several surveys in the wash bottoms, while tracks were not seen affecting the upland populations. In the Dry Morongo Wash area, which had previously very little traffic, tracks were visible in the main wash in the side canyon at the south end of the wash as well as at the northwest end in the area where an unconfirmed seedling had been seen in the past. Little Morongo Canyon, Devils Garden Foothills (one single track, in the main wash), and the Thousand Palms Preserve also had several OHV tracks (an incursion through a fence). Foot traffic was limited where the plants occurred, often in remote areas and on slopes that were not traversable for recreationalists. Wash bottom populations in Mission Creek were in areas where some foot traffic occurs, but the plants were sparse and we did not see them affected. Over the two years of this study, we have noted three plants occurring either in or on the immediate shoulder of an established road: in Devils

Garden last year, and this year in the North Fork of Mission Creek. No specific engineering of banks or flood control was seen to be impacting the species.

Discussion

Federally-endangered triple-ribbed milkvetch (*Astragalus tricarlinatus*) population dynamics, reproductive biology, and ecological relationships are not well understood for several reasons: populations are typically isolated, the plants are cryptic and difficult to detect even under the best circumstances and they typically grow in places that are topographically rugged and difficult to reach (USFWS 2009). As well, the plants documented in wash bottoms have often been single plants in extremely wide and rugged, seasonally-active streambeds, making detection as well as marking/monumentation for long term study very difficult. The goal of this study was to search out more extant populations in the Plan area, and sample genetic material to determine regional population structure, especially relatedness of the Santa Rosa Mountains group.

As stated, many upland populations appeared to be persisting and recruiting new individuals, and we noted more wash-bottom plants than we had in surveys in the past, possibly as a result of flooding in the most recent 2 years, adequate time for seeds to be distributed from upland populations and for seedling emergence. This may be a feature of the dynamics of this species, distributing widely during intense flood events, occupying wash habitat for several years, and then the local range constricting to upland habitat during drought periods. We were not able to confirm this pattern throughout the range with our limited time and the extensive area of potential habitat, however.

Although very few threats were physically apparent to plants seen this year and last, the impacts of temperature increases with climate change, recent droughts, and nitrogen deposition cannot be discounted. Genetic analysis has begun in partnership with the Sachs Lab at UCR, as well as Rancho Santa Ana Botanic Garden and Joshua Tree National Park and this will be essential to answer questions related to conservation genetics.

Recommendations

We recommend continuing support for gathering additional samples and the genetic analysis to understand the fine and coarse-scale genetic structure of these populations. Known populations of the species were sampled from a broad swath of the species' range, following on the 2015 Joshua Tree National Park study (Fraga *et al.*), including those in the Santa Rosa Mountains. This study should provide information on relatedness between upland and wash populations, and some insight into regional dynamics, in addition to the genetic relationship between the Transverse Range and the Peninsular Range populations. In addition, our partners have indicated the importance of symbionts to species like these that occur on poor soils. Symbiotic bacteria occurring in nodules on the roots (rhizobia) may be the key to the species' population or re-population of a given area. Understanding these symbiotic obligations will help determine limitations to species success. We additionally recommend supporting this analysis of rhizobial symbionts.

As has been noted, this species seems to occur on particular soil types, and although soil samples have been collected by various entities, results have not been disseminated (Fraga & Palapil 2012). We sampled soil in areas that have self-sustaining, stable populations as well as ephemeral waif/wash bottom populations and we recommend support to identify the properties of the soils on which triple-ribbed milkvetch occurs.

Ongoing recommendations regarding understanding the biology and life history of the species as relevant to conservation have been enumerated in past reports. However, to repeat these needs briefly, we recommend studying pollination and seed dispersal and demographics. We also recommend supporting ongoing collaboration with rare plant biologists in adjacent Joshua Tree National Park as well as Rancho Santa Ana Botanic Garden in order to share data about triple-ribbed milkvetch occurrence and biology as well as to standardize rare plant monitoring protocols with the aim of providing useful information for effective management.

We recommend continuing to monitor threats noted including the local OHV traffic that is ever-expanding and should be a priority for management of most species within and surrounding conserved areas.

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Appendix IX –
2019-2020 Monitoring Results for the
Little San Bernardino Mountains
Linanthus (*Linanthus maculatus*)
within the Coachella Valley



Coachella Valley Multiple Species Habitat Conservation Plan
& Natural Community Conservation Plan

2019-2020 Monitoring Results for the Little San Bernardino Mountains Linanthus (*Linanthus maculatus*) within the Coachella Valley



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LITTLE SAN BERNARDINO MOUNTAINS LINANTHUS MONITORING

Little San Bernardino Mountains Linanthus (*Linanthus maculatus* (Parish) Milliken *ssp. maculatus*, hereafter Linanthus; Fig. 1) is a small annual herb endemic to southern California. Within the Coachella Valley it is found within the mouth of Dry Morongo Canyon and Big Morongo Canyon near Desert Hot Springs, Little Morongo Canyon, Snow Creek, Whitewater Canyon (UCR CCB 2017, Sanders 2006). Populations also exist on the north and east sides of the San Bernardino Mountains (e.g. Hondo Wash, Homestead Valley and the Bighorn Mountains), east of the Fry Mountains in Upper Johnson Valley, and at the northern edge of Joshua Tree National Park at the mouth of Rattlesnake Canyon and in the Little San Bernardino Mountains; these localities are part of the West Mojave Planning Area (Figure 2). Much of the newly documented range to the north has been discovered in the last 10 years by Rancho Santa Ana Botanic Garden botanists and others. Linanthus is categorized as California Rare Plant Rank 1B.2 (fairly endangered in California and elsewhere, with 20-80% occurrences threatened / moderate degree and immediacy of threat; CNPS 2015).

This species is elusive and little is known about its biology and ecological relationships. During the century following its first collection and description in 1889 only a few populations were discovered. Over the last few decades more populations have been identified and Linanthus habitat has become better understood (Sanders 2006); however, because of the extreme fluctuations in abundance and distribution year to year, more information is needed in order to understand the habitat niche of this species, as well as threats to plants within those microhabitats.

In 2002, a database of historic occurrence records was compiled for all five plant species covered under the Coachella Valley Multiple Species Habitat Conservation Plan (CVMSHCP; Allen et al. 2005). Data were mined querying various herbaria and museums and required considerable effort to remove duplicate points and identify points that were precise enough for geo-referencing. A University of California, Riverside research team then attempted to locate historic occurrence locations on public land for each species and document the existing populations through 500m² vegetation relevés. For Linanthus, only 2 unique historic records occurred on public lands. In 2003 no Linanthus were found at either plot, however in 2004 individuals were observed at one of those plots (n = 1781), and the population was found again in 2005 (n = 2800; Allen et al. 2005). Many new records for the species have been documented since the 2002 study, and we know more about the range of the species within the Plan area and where the species occurs more reliably with adequate rainfall. In 2017, several older occurrences were confirmed to still have the species present (while Linanthus appeared to be undergoing one of its “boom” years). As new locations are still being discovered for the species, an important goal remains to confirm any shift in the range of the species, visiting range edge populations annually.



Figure 1: *Linanthus maculatus* ssp. *maculatus* plant in flower, with scale, taken February 2020 in the Whitewater River Channel, Coachella Valley, CA.

The microhabitat in which it grows is composed of loose, well-aerated sand flats on low sandy benches at the margins of washes, dry canyons and alluvial fans in Sonoran and Mojave Desert scrub and Joshua tree woodland communities at elevations between 195-2075m (CNPS 2015, Sanders 2006) (Fig. 2). To germinate, the species likely requires fine sheet floods that inundate the soil with moisture but do not incise wash channels or erode the sandy topsoil or leave fluvial deposits. It does not occupy substrates with hard surface layers of clay or rock, or loose aeolian sand within and away from washes. On a fine scale, the open microsites this species occupies are absent of shrubs or trees and contain few competing species or dense stands of weedy annuals (UCR CCB 2017, Sanders 2006).

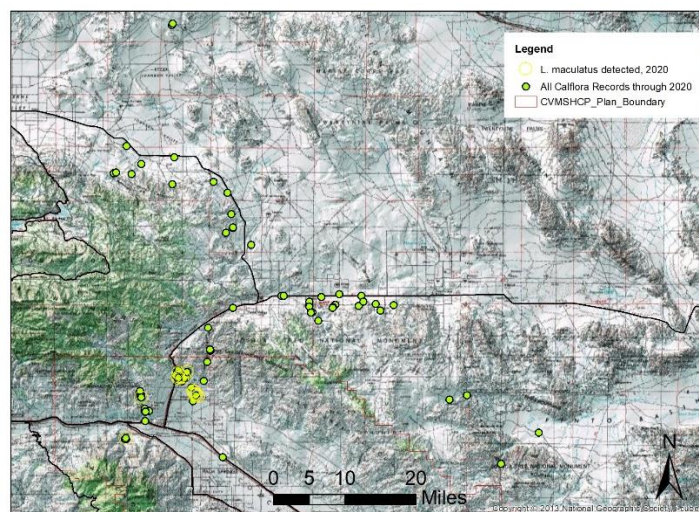


Figure 2: Distribution of *Linanthus maculatus* ssp. *maculatus* records based on georeferenced records available in California, August 2020.

We have employed several approaches to better understand the distribution and abundance of this species. In 2014, an approach was installed to use permanent plots to monitor the species presence, set up at

historic locations of *Linanthus* (see 2014 CVMSCHP annual monitoring report); however, the species was detected at just 3 of them, though it was documented at several adjacent locations. In 2015-2016, *Linanthus* was found to be present at only 2 of the original 12 permanent plots. In order to be more effective in monitoring this small, cryptic species, a more targeted approach was warranted and it was agreed that a better understanding of the factors that influence presence and absence in the habitat is necessary for efficient range-wide monitoring of the species in the future. A newer protocol was established in 2016 using plots with confirmed presence of the species, to looking at microhabitat preferences, species associations, substrate type, and possible competition with invasive species. This protocol was used in the spring of 2016 and again in 2017 with additional plots.

We first documented year-to-year variation in the density of plants present on the plots. Within the seven plots that we monitored in both 2016 and 2017, *Linanthus* showed an increase in all but one plot, with an overall increase of over 22 fold. Native associated species also showed an approximate doubling between the two years, while interestingly, *Schismus* cover did not change significantly. We further documented a possible competitive relationship *between* *Linanthus* and *Schismus* in this microhabitat. We found in 2016 and in 2017 that plots with a high percent cover of *Linanthus* had significantly lower cover of *S. barbatus* and vice versa, although the effect was weaker in 2017 when rainfall was higher.

Although we found a negative correlation between the two species, in order to investigate whether *Schismus* may be impacting *Linanthus*, it is important to rule out microhabitat differences. Soil samples analyzed from *Linanthus* dominated plots showed a mean particle diameter of 837.6 μm (very fine gravelly coarse sand) while samples from *S. barbatus*-dominated plots showed a mean particle size of 613.7 μm (slightly very fine gravelly medium sandy soils that are poorly sorted). Since the sample particle ranges overlap and the means are similar, at this time there is no evidence to suggest that the two species inhabit different soil types, and *S. barbatus* remains a likely threat to *Linanthus* habitat. Additionally, although not detected during our surveys, a new invasive species, stinknet (*Oncosiphon piluliferum*) has been occurring nearby *Linanthus* habitat and may be a threat to the species in the future.

We have made strides in better understanding the microhabitat of this species, and it is reassuring that we have continued to find this species, sometimes in abundance, within the Plan area. Threats to this species include invasive species, climate change, urban development and off-highway vehicle (OHV) recreation, but more work may be necessary to understand how to best manage to conserve this plant. Many of the *Linanthus* occurrences, especially adjacent to developed areas, occurred in areas with light to moderate human foot traffic and OHV use. It is unknown how long these trails have been in use, but we observed many *Linanthus* growing inside tire tracks or on berms caused by OHV's. The openness, lack of large shrubs and absence of channeling that is characteristic of *Linanthus* habitat makes it particularly susceptible to foot traffic, as these are the same conditions that make off trail travel the easiest for both hikers and OHV operators, and so an important question remains whether these two are just correlated in space, or there is a causal relationship. It is unknown if there may be a threshold for disturbance that would cause the decline of the species.

Finally, most aspects of this species' biology, including mode of pollination, dispersal, germination requirements, and seed longevity, remain unknown (Patterson 1989). Most of these items would require in-depth study, as the species' stature is so minute. However, this year, an effort was made to document pollination occurring on the plants.

Objectives

Surveys for *Linanthus* were carried out as part of research and monitoring for the CVMSHCP by the UC Riverside Center for Conservation Biology (CCB). In order to gather more information about threats to the species, we repeated the in-depth 2016-2017 microhabitat surveys, and confirmed presence at several of the transect locations that supported the species in 2017.

Our primary objectives for this monitoring effort were to assess the current abundance and distribution for populations of this species, document habitat attributes and identify potential stressors that may affect its persistence, in particular its tolerance to the presence of *Schismus barbatus*, but including other invasive species, OHV operations and trampling by foot traffic.

Methods

Data Collection

Permission was granted by the University of California, Riverside College of Natural and Agricultural Sciences administration to allow ongoing research during the COVID-19 pandemic. Due to societal and public health restrictions surrounding the COVID-19 pandemic and the impact to staffing and transportation, several sites were not accessed. CCB surveyed *Linanthus* within twelve 10x100 m plots that were selected based upon previous occurrence records along the Mission Creek and Dry Morongo drainages and within. We also sought to confirm persistence for the species in two other locations, following up from the range-wide surveys in 2017, as well as seek out locations to the southeast of the known current extent.

In focused surveys, we resurveyed 2 transects (20 plots) established in 2016 and read in 2017 in order to study the relationship between *S. barbatus* and the density of *Linanthus*. All plots were subdivided into even intervals along the meter line and a plot frame was used for sampling. Plots were run through patches of *Linanthus* of varying density (high- vs. low-density), but of similar habitat (e.g., slope, aspect, associated species, soil surface texture, hydrology characteristics). We then used 1 m² plot frames and record number of *Linanthus* and *S. barbatus*, identified and recorded cover for all species encountered, and recorded slope and aspect from within the frame.

Results

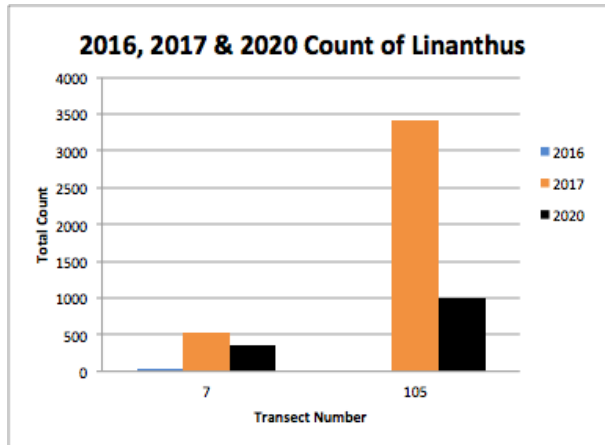


Figure 3. Count of *Linanthus maculatus* plants as a total over 10 plots in two transects for the years 2016, 2017 and 2020.

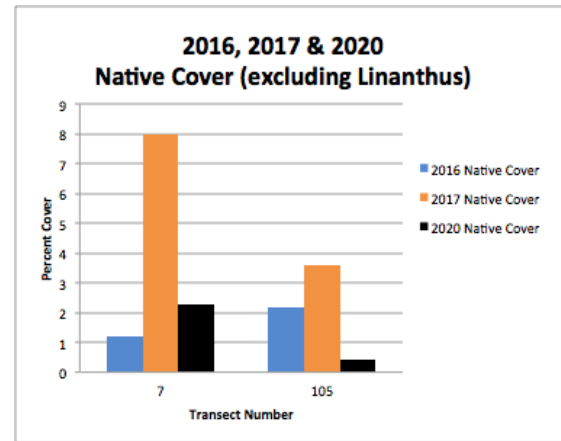


Figure 4. Percent cover of native annual plants (excluding *Linanthus maculatus*) in plots on two transects for the years 2016, 2017 and 2020.

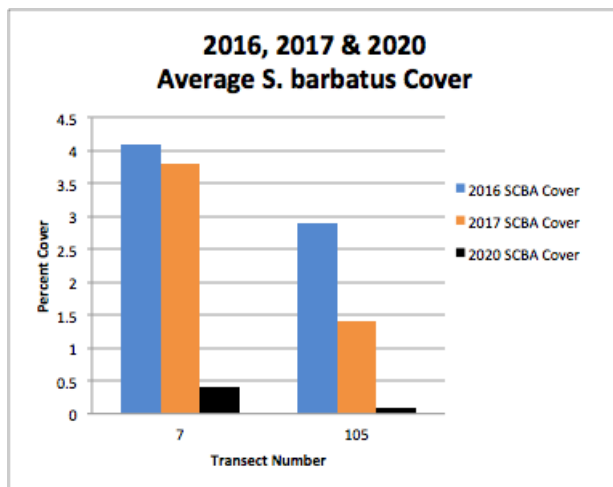


Figure 5. Percent cover of the invasive grass, *Schismus barbatus*, in plots on two transects for the years 2016, 2017 and 2020.

Of the three years studied, the patterns of native annual cover, *Linanthus* count and invasive grass cover did not follow the same pattern (Figs. 3-5). *Linanthus* was most abundant in 2017, as were other native annuals (2016 counts were too low to be visible on this scale; Figs. 3 & 4). *Schismus barbatus* was most abundant in 2016, and decreased slightly in these plots in 2017 (Fig. 5). In all cases, *Linanthus* occurred in open, coarse-sandy microhabitats, beyond the shade of large shrubs.

The relationship between *Linanthus* and *Schismus* was plotted for the years where *Linanthus* was abundant enough to be studied, 2017 and 2020 (Fig. 6). The relationships were generally negative, where

Linanthus counts were highest when Schismus cover was lowest, although there was not a tight relationship found between the two. We did not detect the presence of the new invasive, *Oncosiphon pilulifera* (stinknet).

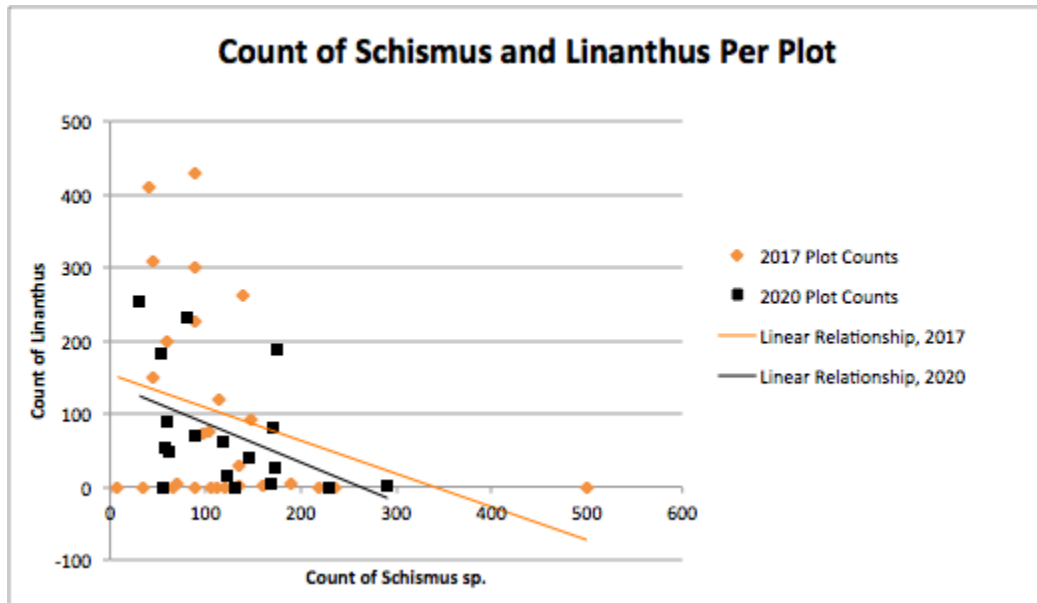


Figure 6. Count of *Linanthus maculatus* and *Schismus barbatus* in plots in 2017 ($R^2 = 0.10$) and 2020 ($R^2 = 0.20$).

Biology and Natural History

This year we recorded an insect visiting and presumably pollinating the plants. We preliminarily identified individuals of the family Anthomyiidae (flower flies or root maggot flies; Fig. 7). The flies were observed systematically seeking out the *Linanthus* flowers, running across the ground from flower to flower, and dipping their head/mouthparts down into the flower. We collected a handful of specimens and found pollen on the underside of the head on multiple flies, presumably pollen from the focal species here, *Linanthus*. We also observed an arthropod, *Collembola* (springtail), on the flowers, but the pollination service of this species was not confirmed.



Figure 7: A fly, possibly of the order Anthomyiidae (flower flies or root maggot flies) visits a *Linanthus maculatus* flower in Desert Hot Springs, CA, February 21, 2020 (Scott Heacox photo).

Extent mapping

During this dry year, we were able to verify that occurrences of *Linanthus* known from 2017 surveys were apparent, both in Snow Creek, Whitewater, Big Morongo (southern edge) and in the Mission Lakes area (Fig. 8). Thus, we did not find an east-west gradient in detection.

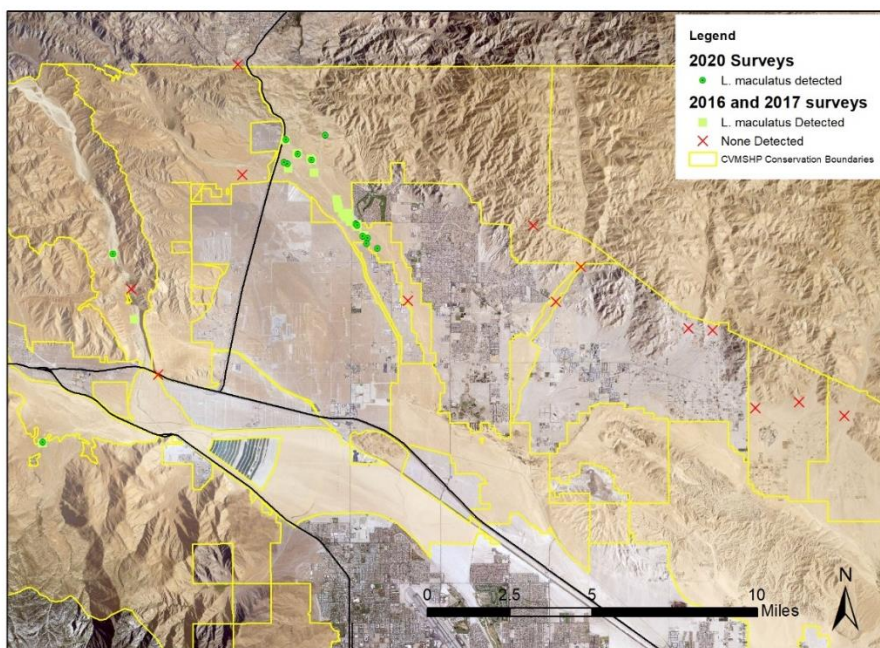


Figure 8: Detections of *Linanthus maculatus* in the west end of the Coachella Valley, CA.

Discussion

Overall, the results of our studies underscored the variability in abundance of this desert annual. This illustrates the difficulties involved in rare plant study, including the difficulty of detection at a useful scale for surveys, the unpredictable window available for detection, and the large variability in time and space for *Linanthus maculatus* populations. However, we found that the species correlates in abundance inter-annually with native annual species. Our surveys resulted in a contribution to the publicly available Calflora database, an update/extension to the intra-regional extent of *Linanthus*. *Linanthus* individuals were found on the same transects as they have been in 2016 and 2017 and we did not detect range contraction, even in the drier 2020.

The association between this plant and the abundance of annual grasses, primarily *Schismus barbatus*, was found to be negative, as in the past. However, this relationship could change across the species range, and plots may need to be rearranged around this hypothesis.

We did not focus this year on the impacts of foot traffic and OHV's, but they were clearly evident at in most sites. *Linanthus* tended to occur in the same gently-sloping well-aerated substrate as also hosted corridors within the desert. It is difficult to tell what the effect of occasional OHV incursion may be in these habitats. As mentioned in previous years, many plants occurred in areas with light to moderate human foot traffic and OHV use. It is unknown how long these trails have been in use, but we observed many *Linanthus* growing inside tire tracks or on berms caused by OHV's. The openness, lack of large shrubs and absence of channeling that is characteristic of *Linanthus* habitat makes it particularly susceptible to foot traffic, as these are the same conditions that make off trail travel the easiest for both hikers and OHV operators, but it is unknown what the plants threshold for disturbance is at this time.

Since so little is known about the natural history, microhabitat requirements, effects of changes in hydrology and tolerance to invasive species of *Linanthus*, it is anticipated that the data collected in our additional study and future surveys, should they be replicable, will elicit more complex relationships to the habitat in the course of additional analysis.

Recommendations

It is recommended that surveys continue on a yearly (or close to yearly) basis to establish the precipitation threshold required for this species to germinate successfully, its tolerance to invasive species and to better understand its current range within the Coachella Valley. Plots with known occurrence locations should continue to be revisited with each future survey effort. Along with tracking recent occurrences, it is also important to study how the range is changing over time by focusing searches on the margins of *Linanthus* known habitat. By revisiting historic plots and suitable microhabitats along the periphery of the historic records and the modeled habitat it will allow the ability to detect if the range is expanding, contracting or even shifting due to various pressures (e.g., development, invasive species, nitrogen deposition, climate change). We also suggest a pollination study and seed dispersal study to find out what factors are responsible for these portions of the plant's lifecycle. We are working in consultation with rare plant biologists in adjacent jurisdictions (e.g. Joshua Tree National Park) in order to share data about *Linanthus* occurrence and biology as well as to standardize rare plant monitoring protocols with the aim of providing useful information for effective management. This information will enable surveys to be timed more effectively, cited appropriately and allow for continued evaluation of OHV recreational activity and invasive species impacts to this species.

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Appendix X – Assessment of the Status of Honey Mesquite

**Assessment of the Status of Honey Mesquite
Stands in the Coachella Valley Multiple
Species Habitat Conservation Plan Area and
Factors Affecting Persistence**



December, 2020

**Prepared by The University of California's Center for Conservation Biology
For The Coachella Valley Conservation Commission**

Assessment of the Status of Honey Mesquite Stands in the Coachella Valley Multiple Species Habitat Conservation Plan Area and Factors Affecting Persistence

December 2020

**Prepared by the
University of California Riverside, Center for Conservation Biology
for
The Coachella Valley Conservation Commission**

Introduction

Southern California's desert populations of honey mesquite (*Prosopis glandulosa* Torr. var. *torreyana* (L. D. Benson) M. C. Johnst.) are an important component of desert biodiversity as key structural components of endemic food webs and appear to be in decline in several areas. These stands occur within diverse biotic associations including desert riparian communities as peripheral to species such as willows, cottonwoods and palms, within and surrounding dry lakebeds, and as isolated sand dune hummocks (Barbour and Major 1977). While honey mesquite are protected both in dune and riparian habitats as components of natural communities under the NCCP portion of the Coachella Valley Multiple Species Habitat Conservation Plan (CVMSHCP), honey mesquite here also provide critical habitat for several CVMSHCP covered species. Species occurring on mesquite sand dunes include the federally-endangered Coachella Valley fringe-toed lizards (*Uma inornata*) and Coachella Valley milkvetch (*Astragalus lentiginosus* var. *coachellae*) as well as Palm Springs pocket mice (*Perognathus longimembris bangsii*), round-tailed ground squirrels (*Spermophilus tereticaudus*), Coachella Valley giant sand treader crickets (*Macrobaenetes valgum*) and LeConte's thrashers, (*Toxostoma lecontei*). In riparian habitats Bell's vireos (*Vireo bellii pusillus*) and Crissal thrashers (*T. crissale*) also utilize honey mesquite. Managing and sustaining honey mesquite is an important objective of the CVMSHCP.

Honey mesquite stands occupy discrete habitats from the west to the eastern end of the Plan Area (Figure 1) and they are declining in some portions of the Coachella Valley, while they appear much healthier elsewhere. There are notable declines in the southern Desert Hot Springs region, while healthier stands occur from Thousand Palms Canyon south to Dos Palmas (UCR-CCB, 2014). There are stands of mesquite skeletons (dead), among a few live trees, on the dunes

of the Coachella Valley National Wildlife Refuge that are of unknown age and include evidence of extensive Native American use. Our objective is to identify the causes of these declines in mesquite, how quickly this is occurring and what may be done to stop these factors from occurring. Further, aside from concerns about currently-noted declines, there are basic questions particular to this community about the age and longevity of stands, and recruitment patterns that need to be addressed for successful mitigation and restoration.

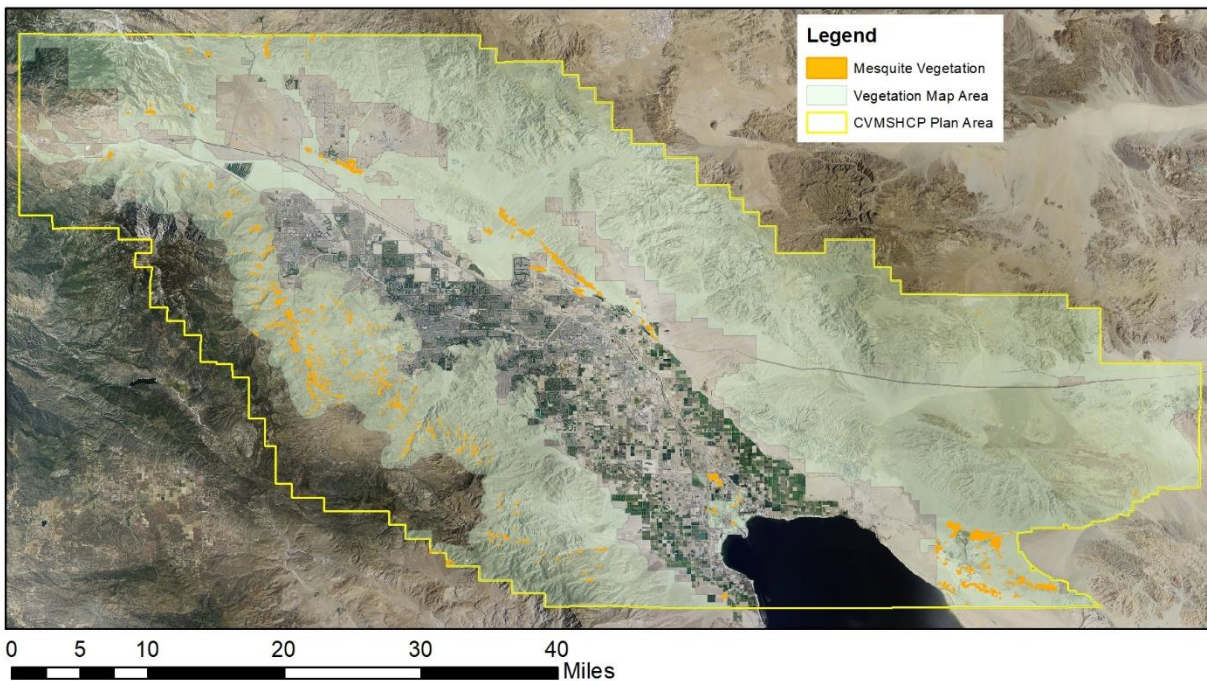


Figure 1: Mesquite stands mapped during vegetation mapping efforts across the CVMSHCP Area (e.g. Menke and Johnson 2015, UCR-CCB 2015-2018; please contact CVCC for data). Mesquite vegetation areas are not to scale- they are buffered significantly in order to be visible.

Understanding the drivers of any declines in the health of a natural community is paramount when developing a restoration plan. Studies investigating the spatial and temporal scale of the problem and the factors involved (abiotic, biotic and anthropogenic) are the basis for a scheme that either directly or indirectly addresses those drivers to prevent further decline and restore the area to the desired state. With the apparent decline of this protected community as a pressing issue, here we provide a summary of our findings and interpretation for developing a more active habitat management plan. Here we focus on three issues relevant to management of this community: 1) documenting the current status of several mesquite stands using modern high-resolution imagery; 2) assessing several potential drivers of mortality through sampling and analysis of existing datasets; 3) review of factors affecting stand persistence and recruitment. Lastly, we close with a summary of our recommendations.

Section I: Documenting Mesquite Community Status and Declines

In 2014, the Center for Conservation Biology at the University of California, Riverside (UCR-CCB) undertook a targeted study to identify trajectories of the spatial and temporal dynamics of mesquite hummock natural communities in the Coachella Valley, identify drivers of these changes, and recommend site characteristics that may support sustainable restoration of mesquite hummocks based on those findings. We documented community dynamics by digitizing the extent of live mesquite visible in a series of historic images, dating as far back as 1938 through 2012. From 1972 to the present only Thousand Palms Canyon and the East End Indio Hills 2 sites were stable or continued to increase though to the present; all other sites showed declines during that period (see UCR-CCB 2014).

In order to get a finer-scale look at patch health which can allow putative factors to be more mechanistically tied to health declines and mortality, in 2019 we captured fine-scale imagery using an FAA authorized UAV flight over 5 focal study mesquite patches (Figure 2). These patches stretched over an approximately 5km linear area along a fault line, from “Fault Line Dunes” (patch 2) to the west, to “Willow Hole” (patches 3 and 4) to the east. The imagery characterized the vigor of the mesquite stands at 5cm resolution (all True Color (Red/Green/Blue) and IR (Red/Green/Near IR) imagery collected at 12 Megapixel resolution in 4:3 (width:height) ratio), captured from April 8 to April 26, 2019, while plants were fully leafed out. NDVI (Normalized Difference Vegetation Index) was calculated as a proxy for health of the plants. NDVI uses the difference in the reflectance of red vs. infrared light to estimate greenness and is commonly used to represent vegetation status (Yengoh et al. 2015). We aimed to characterize the range of health across five stands, and to discover whether the condition of the stand is correlated with the source of water used by the trees.

Focal Mesquite Study Patches in the Coachella Valley

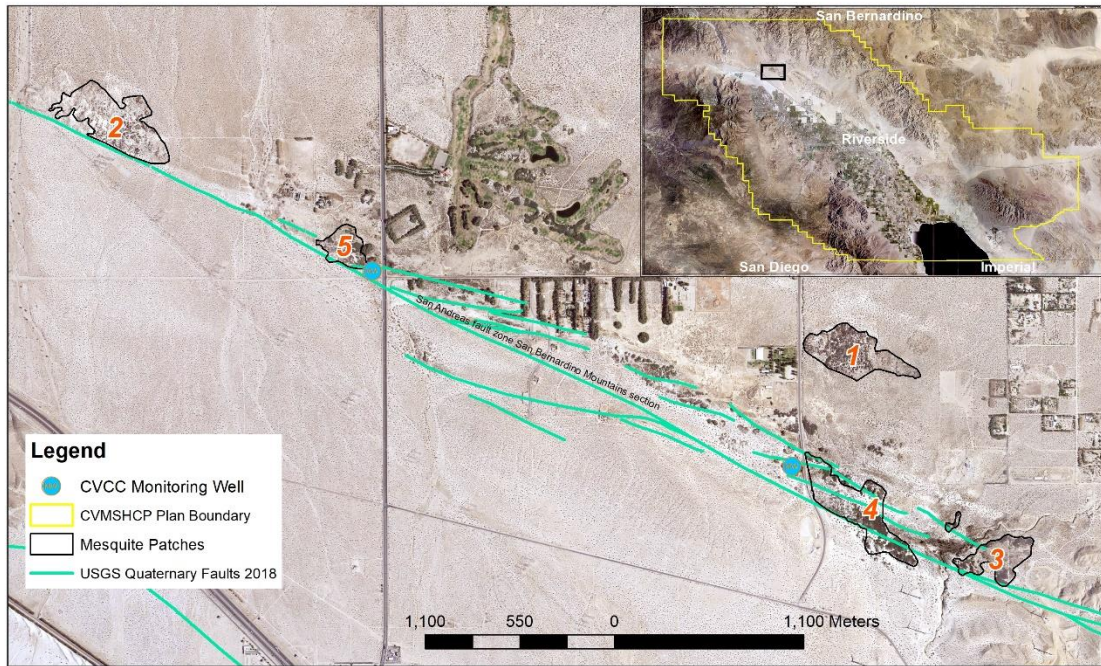


Figure 2: Honey mesquite (*Prosopis glandulosa* var. *torreyana*) study patches within the Coachella Valley, CA. Patches are located along the San Andreas Fault, approximate location indicated on the map (data, USGS 2018). Monitoring wells installed in 2019 by CVCC are also shown.

Patches differed in the range of NDVI values captured, and the amount of area within each category (Figure 3). Most stands had healthy plants interspersed with areas of poor NDVI and dead trees. Patches 1-3 had the lowest values, and patches 4-5 had the higher ratio of healthier mesquite (Figure 3, histogram insets). *Tamarisk aphylla* trees were distinctive in the imagery, showing the highest values in dense patches (Figure 3 patch 5 inset).

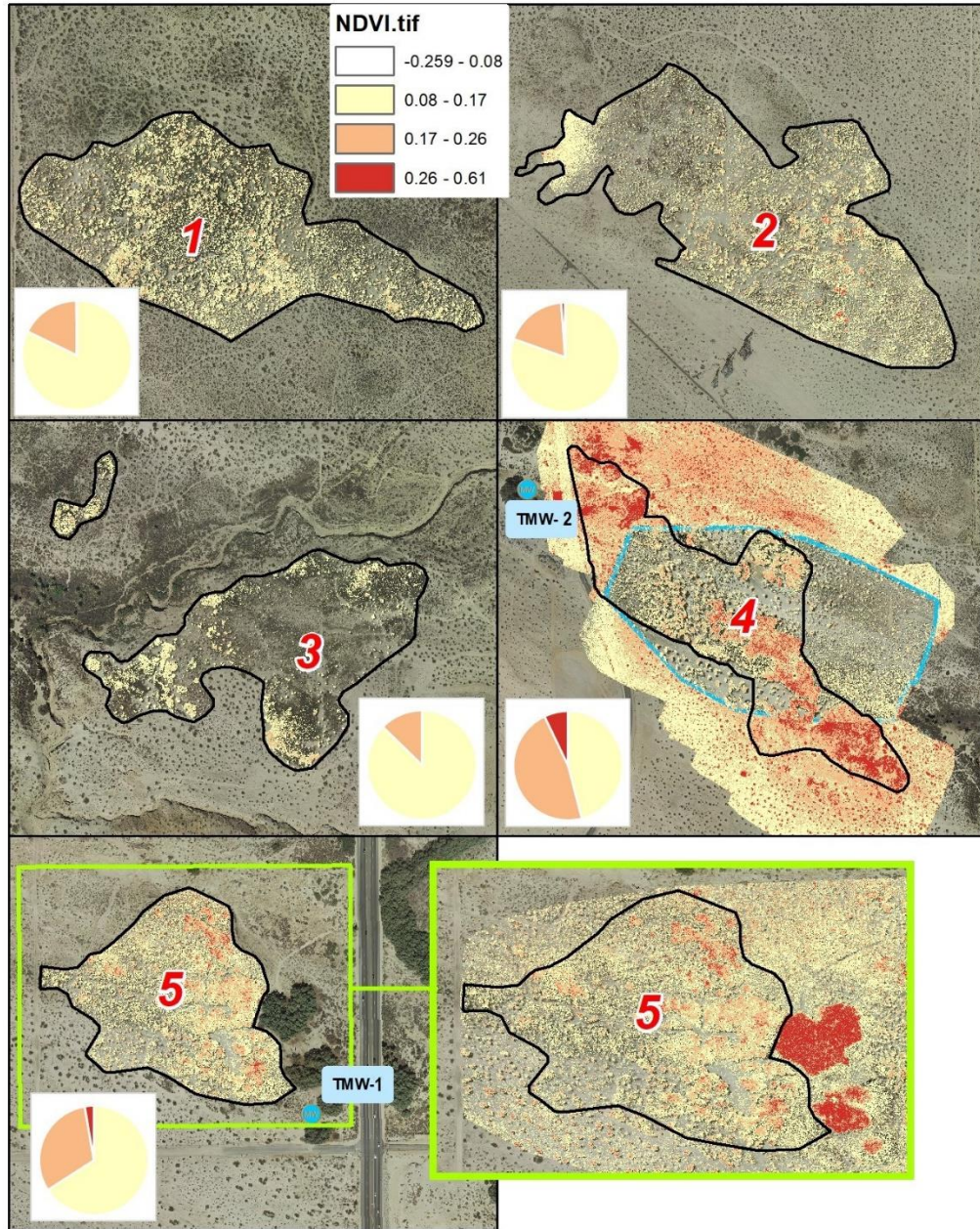


Figure 3: Honey mesquite (*Prosopis glandulosa* var. *torreyana*) study patches within the Coachella Valley, CA, with NDVI values overlain from UAV imagery, spring 2019. Insets show the histogram for the areas outlined in black, with color symbology matching the NDVI image. Mesquite patch extent was heads-up digitized from CVCC-acquired 2015 aerial imagery (shown). The area within the black outlines indicates where imagery was captured to acquisition standards, with the exception of patch 4, for which the area of high quality UAV imagery is outlined in blue. Also indicated in the inset map is the full capture for patch 5, showing the high value (bright red) non-native athel tamarisk trees (*Tamarix aphylla*) on the right side of the frame. Monitoring wells installed in 2019 in association with CVCC are also shown, TMW-1 and TMW-2.

Section II: Putative Causes for Mortality in Mesquite Stands

Physiological Damage by Insects

We conducted preliminary investigations into the possibility that insect pests are a driver of mortality during the summers of 2018 and 2020. In 2018, during unrelated field work at Willow Hole (patch 3 and 4), we noted that some patches of dead mesquite were so heavily infested with wood boring beetles that the grubs' chewing was audibly noticeable even from several meters away. We collected a small amount of dead mesquite twigs in July 2018 and placed them into a terrarium. Soon after, we successfully reared several adult powder-post beetles (*Apatides fortis*, family Bostrichidae) and one metallic wood-boring beetle (*Chrysobothris ca. lateralis*, family Buprestidae). During twig collection, we also swept nearby vegetation with a net and collected a large adult chrysochroine metallic wood-boring beetle, possibly genus *Gyascutus*, off of mesquite (Ueckert and Wright 1974; Ward 1977; Texas A&M AgriLife Extension n.d.). On July 29, 2020 we returned to Willow Hole to collect additional dead branch samples from mesquite patches of varying health. We collected branches from three locations at Willow Hole and swept the mesquite near each branch collection, including the young mesquite found at the northwest side of the area. Sweeping did not yield any possible insect pests, but the branches from one of the collections reared two powder-post beetle morphospecies probably of the genus *Amphicerus* (Bostrichidae) and one small metallic wood-boring beetle, probably of the genus *Acmaeodera* (Ueckert and Wright 1974; Ward 1977; Texas A&M AgriLife Extension n.d.).

It is not surprising to find these beetles in a natural mesquite hummock. These beetles are likely all native to the American Southwest and are therefore a normal part of local mesquite hummock entomofauna. However, more research will be conducted to diagnose exact species to confirm this. Whether these beetles are at a higher density than normal due to climate/drought stress on the mesquite has not been determined. Also, it is likely that at least some of these species preferentially attack dead wood and therefore would not pose a threat to living mesquite, however substantial boring in dead wood can increase the risk of fire (Ueckert & Wright 1974). The biology of many wood-boring beetles is still unknown, so a more thorough examination of infestations of living wood would be necessary to determine threats to living mesquite. Also, wood boring beetles can be quite diverse, so there are certainly many more species that were not sampled, including more metallic wood-boring beetle species, longhorn beetles (family Cerambycidae), death-watch beetles (family Anobiidae), and others. Likewise, there are many other groups of insects that could pose a threat to mesquite, such as seed-eaters, foliage grazers, and fluid-feeders, which were not sampled in this study (Ward 1977).

Groundwater Changes

Plants in desert environments with high LAI (Leaf Area Index: leaf surface area: weight) rely on consistent access to water for survival. Phreatophytes, including many of these species, are plants that require year-round access to water; they live in oases, riparian habitats, and, though seemingly dry, areas where water is at or near the surface due to earthquake faults (David et al. 2007, Catchings et al. 2009). Honey mesquite has been categorized as a facultative phreatophyte (Heitschmidt et al. 1988), indicating that it can shift its primary water uptake between a relatively deep tap root, up to 52 m recorded (Rundel 1963), to more shallow near-surface roots that take in water from rainfall events. This designation may be misleading, suggesting that honey mesquite can thrive either with a high water table, or with the infrequent desert rain typical of the Coachella Valley. The common denominator of the different abiotic associations where honey mesquite can thrive is that they are restricted to areas of very high water tables, or temporarily high water tables following uncommon summer monsoon floods where seedling honey mesquite germinate and extend their taproots to maintain contact with groundwater levels. While mesquite can absorb surface rain water through near-surface roots, all long-lived honey mesquite stands in the Coachella Valley occur where there is a relatively high water table, often associated with earthquake fault zones.

The 2014 study by UCR-CCB utilized ground-penetrating radar along the surface of the dunes to assess the depth to water at several sites. Findings showed probably near-surface water at a more vigorous stand, but at all other sites, there was no detectable water from the surface (UCR-CCB 2014). As well, the regional well records studied showed increases in depth-to-water, though a lack of access to finer-scale well data precluded the specific attribution of the decline of some stands vs. others due to declines in the ground water table. Sites without access to near surface ground water showed the greatest declines in live mesquite.

In August, 2018, two wells were established by CVCC in conjunction with Coachella Valley Municipal Water District: one along Palm Drive (Figure 2; Figure 3 panel 5, “TMW-1”) and one in proximity to the Willow Hole patch 4 (Figure 2; Figure 3 panel 4, “TMW-2”). Due to the short duration of the time series available as of this writing, no particular trend was apparent, although large (approximately 4%) rise was recorded in the week from February 14th to February 21st for TMW-1, likely following the large storm event on February 14, 2019 (Figure 4). The other well, TMW-2, also displayed a rise during this period, but it was part of a likely seasonal trend of increased water availability starting in November.

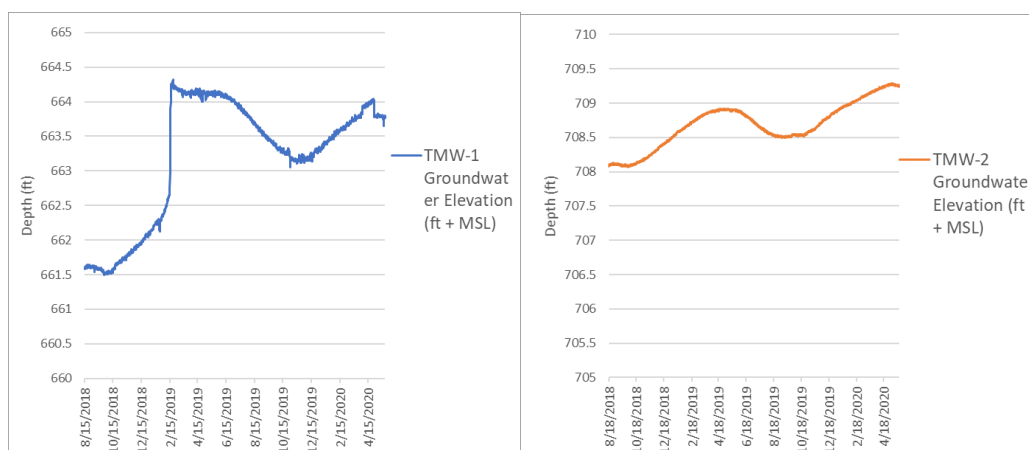


Figure 4: Well depth data provided by CVCC for two wells established within the study area in the Coachella Valley, CA. TMW-1 is located to the west of study patch 5 and TMW-2 is near study patch 4. Shown are data for August 2018 to May 2020. The water level data was converted to groundwater elevation by taking the surface elevation minus the transducer depth below ground surface, and adding the transducer reading.

Regionally, peer-reviewed literature indicates ongoing groundwater depletion in neighboring groundwater basins such as the Whitewater groundwater basin (Thomas & Famiglietti 2015). No peer-reviewed summary of the regional trends within the Mission Creek sub-basin were located, and a hydrologic study is beyond the scope of this analysis. However, the 2014 study by UCR-CCB analyzed records provided by CVWD for wells in near-proximity to this site and results are summarized there, reporting some declines in mean ground surface to aquifer depth in four of the six wells studied (see UCR-CCB 2014).

Water Use Differences in Healthy vs. Unhealthy Mesquite and Competition from Other Species

The ratios of stable isotopes of hydrogen (^2H or deuterium (D) relative to ^1H ; hereafter δD) and oxygen (^{16}O relative to ^{18}O ; hereafter δO) in water have been used to establish patterns of water use in plants by using stem water to characterize the water that was drawn up by the roots (Ehleringer and Dawson, 1992). Although the isotopes themselves are stable, there is a change in the *ratios* of one isotope to another for each element as processes such as evaporation and condensation occur, as the “lighter” isotopes of each element evaporate more readily than the “heavier” isotopes, creating a shift toward heavier isotopes in the remaining, non-evaporated water (Dawson and Ehleringer 1998). In practice, this gives us the ability to identify plants that are using shallower, more evaporated water, as opposed to deeper, less-evaporated water, assuming the sources of these waters are not different.

As mentioned, plants with access to more secure water as shown by GPR in the 2014 study were assessed to be healthier. The current study built on this idea, tying specifically the depth of the water being used by the plants to mesquite health. As described in Section I, NDVI values acquired from aerial imagery capture give us an idea of the approximate greenness of the patch, which here we use as a proxy for mesquite condition/health. We used the 2019 NDVI maps to divide the five mesquite focal study patches into low, medium and high NDVI (\approx “condition”) categories for sampling stem water, and sampled three ~ 1.5 cm diameter stems per patch from each of the categories (from different trees, as far as could be discerned on the ground). Several patches did not have all three condition categories available for study, however. As well, since stem water can only reliably represent groundwater where it is not evaporated prior to conduction to the stem, only stems lacking greenness or those that were covered in a cork layer more or less completely, and minimally green were sampled. We additionally sampled three stems from co-occurring shrubs (*Atriplex* species) per patch, as well as a total of three samples of *Tamarisk aphylla* that were growing adjacent to two of the patches. Stems were collected and immediately transferred into 10ml BD Vacutainer serum blood collection tubes (no additives), sealed with Parafilm, frozen and transferred to the lab for water extraction and analysis. For analysis, we included six mesquite stems per patch, three from the highest category and three from the lowest category per patch (since all patches did not have three categories for comparison). Stem water was extracted using standard methodology in a vacuum line in the lab of Professor Louis Santiago at UCR and subsequently sent to the UCR-CCB FIRMS laboratory (ccb.ucr.edu/facilities/firms) for analysis. Additional methodological details are available upon request.

We performed a comparison of the stable isotope values, δO and δD , in mesquite stem water across the stratified sample of stems. For analysis, we included patches 2, 3, 4 and 5, since patch 1 was set apart from the others away from the fault line (potentially different hydrology), and had a lack of medium-to-high areas for comparison. NDVI values were extracted from the UAV-acquired map, using the highest value of NDVI for the tree within a radius of 1.5m from the original GPS point during stem collection, to account for GPS error. We focused on δO , since the signature change from evaporation of waters tends to be more pronounced for this component within our xeric environment. We performed linear regression in R (R Core Team, 2020), relating δO to NDVI.

We found that there was a significant negative relationship between δO and NDVI ($p=0.006$, adjusted R-squared $=0.265$) without accounting for inter-patch differences, and an overall improvement of the model prediction accounting for patch ($p=0.003$; adjusted R-squared 0.469). When looking at the average differences between low and high-NDVI over both isotopes, the directional trend towards evaporated water is apparent in all four patches (Figure 5). As well, even without accounting for patch differences, using univariate analysis, the Low condition patches were near-significantly higher in δO than the High condition patches ($p=0.055$, Figure

6). The patterns seen here likely mean that the water used by the healthy mesquite originates in less-evaporated (deeper) water than the water accessed by the mesquite that is doing poorly.

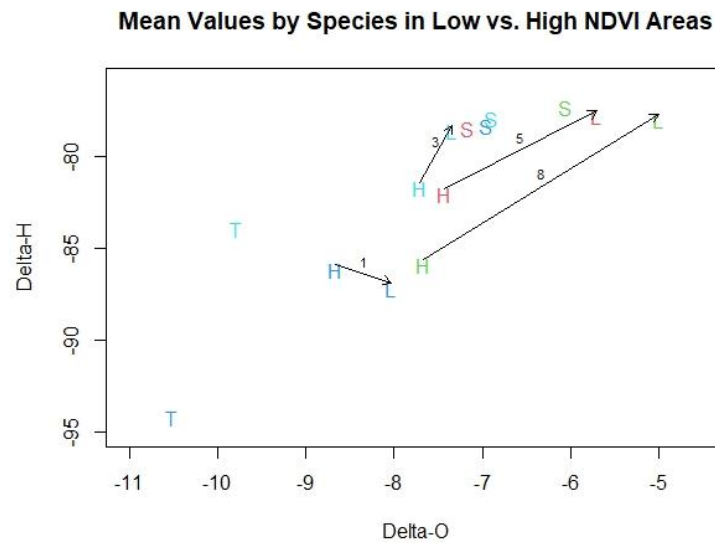


Figure 5: Water stable isotope values for stems of mesquite in High (H) and Low (L) NDVI areas within each focal patch (represented by color). Arrows represent the change in values from High to Low in mesquite. Also shown are average values for shrubs (S) and raw values for athel tamarisk (T).

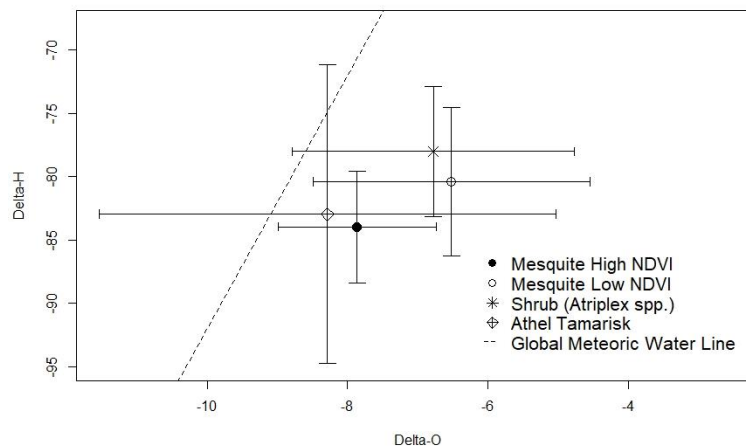


Figure 6: Water stable isotope values for stems of mesquite in High and Low NDVI areas for patches 2-5, averaged over all values from the four patches (n=48), average values for shrubs (n=12) and tamarisk (n=3). Error bars are one Standard Deviation.

Competition from Non-Native Species

Mesquite is subject to the competitive environment of these areas where water is available at otherwise dry landscapes. In particular, non-native athel tamarisk can form massive stands and the trees have been planted near and around private properties for shade and wind-

breaks and also alongside infrastructure for soil erosion control across the U.S. desert Southwest (DiTomaso et al. 2013). While it does not reproduce via seed commonly, it can reproduce vegetatively, and its massive vegetative biomass can pose a management challenge. In addition, it is thought to be a competitor with native species such as mesquite, reducing underground water availability. In the study area, this species is also highly visible on the NDVI maps due to its large volume of leaves (Figure 3, panel 5 inset), and it is known to transpire large quantities of water (DiTomaso et al. 2013). We found that tamarisk is accessing the deep, reliable water more than even healthy mesquite. Two of the three tamarisk trees measured showed the least, and third-least-evaporated values of all plant samples, respectively.

This species has justifiably been a target of invasive species control, and CVCC implemented a large control effort in 2018 coinciding with the installation of wells TMW-1 and TMW-2. Though we could not directly detect a competitive interaction between these species and a release following control, it is likely that water use by tamarisk would negatively affect co-occurring mesquite. As well, as a driving overall factor explaining declines, the pattern of tamarisk does not completely coincide with observed mortality of patches.

Detecting competition by established stands of plants against a background of seasonal water fluctuations, ongoing groundwater declines and other environmental changes is extremely difficult and would probably need a more systematic study design. Set within a very large integrated area, it may be that effects of such reductions in invasive tamarisk may be localized. Even if the effects of control efforts of this species are not apparent now, a longer record and more extensive control effort may display reductions in groundwater declines and an improvement in mesquite condition. As well, as the desire is to be effective in benefiting these relatively small, localized areas of conservation interest, localized control efforts may be adequate.

Other Factors

There are several other factors that we did not assess in this study. We did not investigate pathology as a factor in declines, and this could be investigated. As well, climate change-induced rising temperatures may be impacting these stands, however, it is unlikely that rising temperatures would solely explain declines in a desert-adapted phreatophytic species.

As for the possibility that these stands are in decline due to regional drought and vulnerability—the southwestern US is experiencing aridification, especially since 1999 (Overpeck and Udall, 2020). Therefore, a lack of availability of seasonal precipitation, supplementary water as well as groundwater declines could be tied to drought. However, if this were the only factor, we would expect to also see strong declines in other species that are more directly affected by rainfall patterns, such as *Atriplex* species. While some mortality of shrubs was noted in the 2019 vegetation mapping effort within the Coachella valley (UCR-CCB 2018), we did not note extensive mortality of shrubs in this area. Still, if groundwater sources are

declining to where stands are unable to reach deep sources, a decrease in precipitation would further exacerbate declines. As we have shown here, those mesquite currently using primarily shallower soil water in our study were less healthy.

In terms of the impacts of off-highway (OHV) vehicle use, UCR-CCB investigated this issue as part of the 2014 study. It was found at the time that off-road vehicle trails were not more abundant adjacent to dead or stressed mesquite compared with more vigorous mesquite patches. Looking at the patterns of mortality and the lack of evidence of OHV's impacting areas these specific areas, particularly in the central part of patch 4, it would not seem likely that physical damage from these vehicles is primary in mortality. Further, these vehicles tend to occupy the "path of least resistance," not commonly taking a path through rough, thorny vegetation, but keeping to the paths in the sand.

Section III: Stand Persistence and Recruitment

Stand Size and Age

In areas with active wind-blown sand movement, mesquite hummocks provide structure that can be a nucleus for accumulating and stabilizing large quantities of aeolian sand. With access to sufficient water, mesquite can maintain high enough growth rates to continue to stay above the dune sand accretion. This can result in large dunes several tens of meters tall with living mesquite emergent throughout. It also can give a misleading impression that the mesquite germinated and grew on the dune; rather the dune grew around the mesquite.

Without a precise geo-chronology, an extremely-long-term record of dune topography, we cannot definitively establish how tall the trees are, how old they are, and at what level topographically they in fact germinated. However, another study on this species on dunes in Jornada Experimental Range (New Mexico, USA) excavated dunes and provides some guidance. Similar to our environment, mesquite plants at Jornada acted as foci for the establishment of dunes, but by contrast this process was thought to be due to land disturbance and the erosion of the soil from surrounding areas, that then established around the plants. Researchers found that the crown of the main trunk itself originated at approximately the level of the surrounding ground outside the dune, and that the age of the trees (estimated using ring counts) correlated strongly with the volume of the crown and with the width of the dune itself (Gadzia and Ludwig, 1983). At several of our sites, the masses of branches are so thick and continuous that it is hard to designate a single tree; in fact, Gadzia and Ludwig did similarly locate several trees growing together at one site, and this was not apparent until after excavation.

If we apply Gadzia and Ludwig's (1983) methodology to estimate ages for our stands, apparent canopy volumes at our sites tend to be an order of magnitude larger than those represented in the Jornada study (>5m height, >10m radius), and result in suspect age estimates of 1000+ years using their volume formula and age estimate linear regression (see Gadzia and

Ludwig, 1983 Figures 2 & 3 for equations). Therefore, it's likely that we cannot assume similar growth rates for mesquite trees within the Coachella Valley; however, the concept could be applied if it was possible to sample cross-sections of crowns, for example, at sites that have already suffered mortality. At the Jornada Site, they generally found that older plants were located towards the center of the dune and younger at the edges, as would be expected. Crown width pattern was not investigated here, but individual crowns are difficult to distinguish among the center patch, ~100 m across (Figure 7 center patch), and this may be composed of various individuals.

The Jornada study also documented cultural artifacts on the study transects, noting positioning relative to the dune sands and trees (Gadzia and Ludwig, 1983). While their interpretation of these patterns would not hold within our study area (our mesquite trees are positioned in space along fault water, not in a grassland, so the patterns of cultural use and the progression of dune accretion in space is likely very different), further study should incorporate traditional ecological knowledge and this type of cultural evidence. This may provide the needed information about the conditions that existed when these stands originated, if we desire to replicate conditions that once again favor recruitment.

Though populations of long-lived species such as these can persist with low recruitment rates (Schweiger et al. 2020), simply replacing individuals slowly over time, a lack of recruitment of new trees at even that very low necessary rate may be another factor leading to the decline and disappearance of patches. In order to persist, there needs to be an environment that allows seeds to disperse to a site, germinate and grow, or alternatively, for trees to reproduce via cloning, or vegetative reproduction. Some documentation was found concerning vegetative reproduction in a search of literature for *Prosopis* species, meaning that trees can sprout new trunks as a mechanism to persist in addition to reproduction by seed (Fisher et al. 1946). Still, some trees will naturally suffer mortality due to hydraulic failure, or inability to meet a carbon balance (Sevanto et al. 2014) if growth of roots is not also renewed with sprouting.

Recruitment

There is some research that has focused on other *Prosopis* species that is relevant and useful to incorporate here. Observations of *Prosopis glandulosa* var. *glandulosa* Torr., a sister variety, have similarly found that tree recruits are very rarely observed underneath the canopy of adults (Whittaker et al. 1979; Archer et al. 1988). A physiological study to explain their lack of co-occurrences with adults in grasslands showed that reduced light levels and competition from herbaceous plants (grasses) both contributed to lack of survival of mesquite recruits (Bush & Auken 1990). In terms of seed limitation (inadequate number of seeds produced by stands for sustaining populations), although multiple seed predators inhabit the mesquite hummocks including the round-tailed ground squirrel and other rodents and insects (Ward 1977), there is no evidence that this is limiting to establishment of recruits. There appears to be at least some, if patchy, flowering and fruit at some patches each year, and we have not observed a lack of

pollination and seed set on the whole that would seem to explain the lack of recruitment (C. Barrows, pers. obs.). As mentioned, it is not possible without excavation and further study, to determine to what degree vegetative reproduction is occurring in this system, where the crown is partially buried within the dunes.

The only significant seedling recruitment of mesquite noted during the study period was on the north side of Willow Hole, patch 4, where we noted 8 sapling-to-adult stage recruits ranging from 1-2 m in height, and 3-7 cm in diameter at ground height (Figures 7 & 8). The area was also the site of control of *Tamarix* species by CVCC, primarily aimed at athel tamarisk (*Tamarisk aphylla*), but saltcedar (*Tamarix ramosissima*) was also present. This athel removal was the last of a series of removal efforts on the site, over approximately 10 years (C. Barrows, pers. obs.). The previous efforts removed a number of large saltcedars, leaving the athel tamarisk until the recent effort. Considering the diameter of the main trunk of the mesquite recruits, it is likely that these trees may have established and benefited from competitive release due to these previous efforts. This area was along a flow course as evidenced by the silty surface substrate, and visually at the original height of the land prior to dune development.

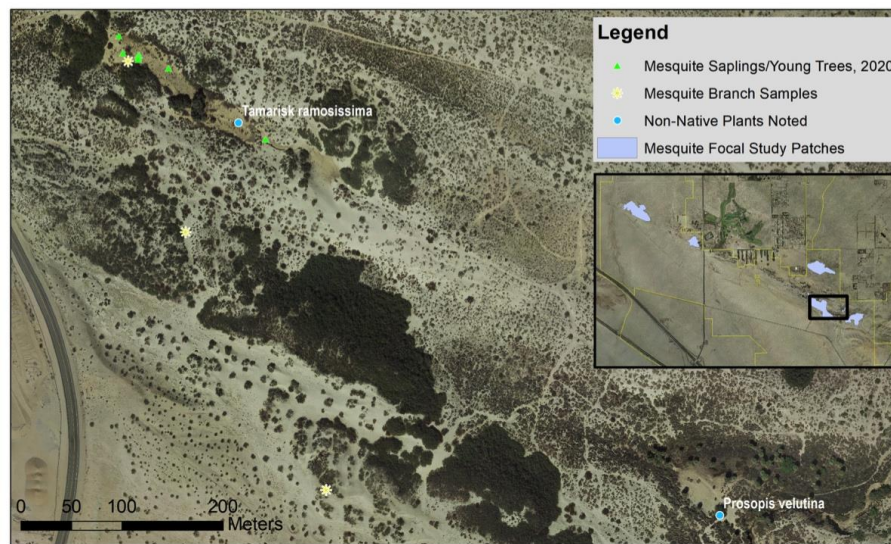


Figure 7: Mesquite recruitment in a silty area on the north side of the Willow Hole stand (4), coincident with tamarisk control efforts by CVCC. Also shown are the other non-native species recorded in the area, the invasive saltcedar (*Tamarix ramosissima*) and velvet mesquite (*Prosopis velutina*) as well as the locations of the 2020 branch collections for studying insects occurring on the plants.



Figure 8: Mesquite recruitment in a silty area on the north side of the Willow Hole stand (4), coincident with tamarisk control efforts (Left), and one of the recruits, approximately 2 m in height and 7 cm in diameter at ground height.

Summary of Historic Persistence

Based on the factors described above and the situation of the trees relative to groundwater, along a linear path, the long-term *natural* spatial patterning most likely involved 1) recruitment at the land surface level near fault-water; 2) accretion and growth of the dune and associated growth of the mesquite branches outwards from the dune, along with possible vegetative sprouting from the crown; and 3) age-related mortality (though it is unknown what the maximum age of the trees is), and erosion of sand. As stated, the age of these trees, and the maximum possible age of this species is unknown, but trees of the sister variety are recorded to live more than 100 years (Miller et al. 2001), and it is likely that the current mortality and lack of recruitment seen is not natural, due these effects occurring simultaneously. In terms of seedling establishment conditions for mesquite dune hummocks: for recruitment to occur, a low-lying (non-accreted) area with proximity to groundwater must be available. This occurs locally with larger flood events, infrequently due to summer rainfall, but also with winter rainfall, and this flooding would additionally add the benefit of scarification that improves germination of hard-coated leguminous species. These energetic floods move through the valley, causing erosion and movement of soil. With patches of mesquite situated in a line along the fault, running nearly perpendicular to the downhill energy of moving water, lower-lying areas and breaks within the line of dunes may experience erosion of sands and depositing silt. At various locations along the faultline, these would provide the new foci for growth of young mesquite while the older trees continue to grow until they die. Presumably, when an older patch dies, a cycle of decomposition, wind erosion and then stochastic flood erosion may begin the patch anew. This hypothesized naturally sustained cycle needs further evidence for substantiation, but may in the meantime be considered for maintenance of the communities into the future, even if age-related mortality and recruitment limitation are not the only factors involved.

Section IV: Restoration Summary Recommendations

Review of Putative Factors Causing Decline

First, we review the factors that may be contributing to decline that were considered here, their likelihood for contributing to these observed declines and how they may be mitigated. Recommended in association with any restoration planning or active site management would be to continue to monitor these and any other potential drivers of mortality. This information is necessary to further disentangle the importance of each of these in natural/historic dune persistence and in the face of any more recent drivers of mortality. The presence of insects that we detected seems to be mostly in line with natural patterns, or likely to be a symptom rather than a cause of decline in these stands, and so no cause to currently mitigate this factor, although continued monitoring of the stands for both insect pests and pathogens is recommended. Groundwater changes have been identified as a likely causal factor in another species of mesquite, screwbean mesquite (*Prosopis pubescens* Benth) (Foldi, 2014), and based on the relationship seen between a lack of deep water access and mesquite health, this may be among the leading contenders to explain the mortality seen here. Due to the timing of the tamarisk treatment and well installation in the Willow Hole Conservation Area, we were unable to analyze more definitively the relationship between tamarisk presence and groundwater depth as a cause of decline in mesquite. However, the elimination of competition with non-native species, seen to be accessing and using deep water sources, continues to be a recommended way to benefit stand health, at least in localized areas, even if they are not the main driver of declines. Thus, continued monitoring of the wells adjacent to the mesquite stands, control of competitive species such as tamarisk, and compilation of a regional dataset for wells, in coordination with local water agencies would all be recommended.

Further Study

To inform specifically the tactics for active restoration, answers are needed to persistent questions about the historic distribution, natural or anthropogenic maintenance of the system, and the progression of the community. First, since there is evidence of cultural use of at least the Willow Hole area and other sites, it may be beneficial to access available Traditional Ecological Knowledge (TEK) from local experts. It has also been shown that many California plant communities that were formerly thought by Westerners to have established without human intervention were anthropogenic, created by native people for community, sustenance and other traditional reasons (Anderson et al. 2013). It may be useful to incorporate evidence from the spatial distribution of pottery sherds on the landscape, similar to Gadzia and Ludwig (1983). Information such as any history of cultivation or other active management, and the distribution of trees and dunes over time would be beneficial. This would be useful in association with, as mentioned, aging the stands to understand the contribution of vegetative sprouting to persistence (i.e. are local trees producing new sprouts through time), trajectory of age and mortality locally.

This may be accomplished as suggested, via excavation of the crown of trees that have already suffered mortality, using ring-counts, and examining the bases for sprouting. Second, ongoing mapping of mesquite stands should continue with regularity, to ensure detection of living coverage extent and mortality.

Information for Restoration

Relevant to restoring mesquite stands with this holistic view, we may consider that if we cannot slow or stop mortality in stands, that these will need to be replenished with recruitment of new trees. In addition, new recruitment and establishment may need to occur in different/likely adjacent areas than currently support mesquite. Areas that are on the northward side of the current hummocks (such as in Figure 7 & 8) where surface flow is blocked, and there is a concentration/infiltration of water may be amenable to recruitment. Or, areas where flooding has penetrated the line of dunes and laid silt should be managed as potential new stands, and may be the most obvious locations for active invasive plant control, tree planting and restoration. Areas such as these could be detected and mapped by looking at and mapping seasonal water flow courses and monitoring soil moisture at various sites post-precipitation event. Secondly, seed input may be key to allowing for passive restoration via high propagule pressure (i.e. large numbers of seeds coming in from mature stands). Where mesquite stands have declined, reinvigoration of the dune may not occur without active seed input/application or planting. In all cases, potential restoration sites are recommended to include near-surface groundwater and a stable aquifer in order to provide a sustainable mesquite hummock community. For planting, seasonally, this may need to coincide with the highest natural soil moisture levels, which may be the early spring season (prior to flower development and seed fall). The species has shown plasticity in this respect; seedlings in Texas were observed to emerge both in fall and spring due to bimodal rainfall (Brown and Archer 1987).

Following restoration, management plans should be implemented that assume the same process of dune accretion may occur in these new areas, assuming that the sand-flow corridors as protected by the Plan remain in place. Thus, we may expect large dunes to develop in these areas, and a proactive approach to ensure the integrity of infrastructure would be wise.

Moving forward, in line with Plan actions to monitor hydrologic regimes and correlate these with mesquite hummock health, here we proposed a plan for continued research, monitoring and restoration. The sections described above summarize the status of the mesquite stands in the focal study area, probable causes of decline, and how age and recruitment may be occurring in these stands. While we cannot definitively identify groundwater declines as the primary cause of mortality, our conclusions here help establish background for restoration of the site, based on the studies that were performed.

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Appendix XI –
2020 Coachella Cowbirds-
Riparian Birds report



Riparian Bird Surveys and Brown-headed Cowbird Management in the Coachella Valley 2020

February 4, 2021

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Introduction

The Coachella Valley Multiple Species Habitat Conservation Plan (CVMSHCP 2007) identifies five species of riparian birds as targets for conservation, the Willow Flycatcher, Least Bell's Vireo, Yellow-breasted Chat, Yellow Warbler, and Summer Tanager. Additionally, the plan covers Crissal Thrasher, which inhabits both dense mesquite scrub and riparian habitat, and identifies the Brown-headed Cowbird as a potential threat with management concern (Table 1). In order to conserve and manage these species, CVMSHCP program goals include the conservation of 3,870 acres of the Coachella Valley Stormwater Channel and Delta Conservation Area, the permanent establishment of 44 acres of Sonoran cottonwood-willow riparian forest, and the conservation of at least 781 acres of core habitat for Crissal Thrasher.

Table 1. Riparian bird species identified by the CVMSHCP for conservation monitoring.

Common name	Code	Scientific name	Status
Willow Flycatcher, incl. ssp. Southwestern Willow Flycatcher	WIFL	<i>Empidonax traillii</i> (<i>Empidonax traillii extimus</i>)	State Endangered (Federally Endangered)
Least Bell's Vireo	LBVI	<i>Vireo bellii pusillus</i>	State Endangered/ Federally Endangered
Yellow Warbler	YEWA	<i>Setophaga petechia</i>	State Species of Special Concern
Yellow-breasted Chat	YBCH	<i>Icteria virens</i>	State Species of Special Concern
Summer Tanager	SUTA	<i>Piranga rubra</i>	State Species of Special Concern
Crissal Thrasher	CRTH	<i>Toxostoma crissale</i>	State Species of Special Concern
Brown-headed Cowbird	BHCO	<i>Molothrus ater</i>	None (potential threat)

From 2002 to 2004, the Center for Conservation Biology conducted baseline surveys for these riparian bird species and established standardized monitoring survey protocols (Allen et al. 2005). The baseline surveys covered 18 riparian sites in the Coachella Valley with a total of 116 count points.

In 2014, the San Diego Natural History Museum (SDNHM) performed repeat surveys at seven of these sites that were identified as higher priority on the basis of presence of target species from 2002-2004 and lack of recent surveys. The 2014 resurvey found low numbers of target riparian bird species compared to 2002-2004 levels in the Coachella Valley, and in comparison to neighbouring regions, such as Anza-Borrego Desert State Park. The study also found high numbers of Brown-headed Cowbirds, with 100% nest parasitism of the Least Bell's Vireo at Chino Canyon (Hargrove et al. 2014). Three sites, Chino Canyon, Dos Palmas Preserve, and Whitewater Channel, were identified as having the most potential for

riparian bird habitat where cowbirds were likely depressing riparian bird populations below a sustainable level, thereby creating a population “sink.” Therefore, at least three years of cowbird control was recommended in conjunction with nest monitoring. Broader-scale monitoring of population trends that includes additional riparian sites was recommended at a five-year interval.

In 2017, Cowbird control was implemented at the two sites where access was granted, Whitewater Channel, and Dos Palmas Preserve (San Diego Natural History Museum 2017). Two traps were placed at each site, and 75 cowbirds were captured at Whitewater Channel while nine were captured at Dos Palmas Preserve, using modified Australian Crow traps (Griffith and Griffith 2004). Due to the low number of cowbirds captured at Dos Palmas Preserve, alternative methods of cowbird capture were recommended, namely targeted mist-netting.

In 2018, 55 Brown-headed Cowbirds were trapped in the two Whitewater Channel traps (San Diego Natural History Museum 2018). In 2019, four traps were established in the Whitewater Channel, and a total of 79 cowbirds were trapped (San Diego Natural History Museum 2019a).

This report summarizes trapping efforts in the Coachella Valley in 2020. Cowbird removals at Dos Palmas Preserve have been discontinued due to low capture rates, and four traps were placed in the vicinity of the Whitewater Channel, in the same locations as in 2019.

In an effort to monitor sensitive bird species distributions along the Whitewater Channel in off years between standardized point count monitoring, funding was provided to perform several scouting surveys along the length of the channel. Beginning at the wastewater treatment plant in Indio, treated effluent and agricultural and urban runoff create a continually flowing stream for eighteen miles down to the Salton Sea. Vegetation throughout this stretch of channel varies from recently cleared short-statured wetland vegetation to 30-foot tall riparian forest. Survey priority was focused on sensitive species, though all bird species observed were recorded. Additionally habitat conditions were noted through the length of the channel.

Methods

Cowbird Trapping

Four cowbird traps were installed and opened on 21-22 May 2020, all in the general vicinity of Mecca and near the Whitewater Channel (Table 2, Figure 1). All traps were checked and maintained on a daily basis, and were labelled with signage (Figure 2). Traps were shut down between 8 July and 20 August depending on productivity.

Table 2. Locations of four cowbird traps in the Coachella Valley, 2020.

Trap	Latitude	Longitude	Dates in operation
Whitewater Delta Trap #1 (WW1)	33.512734	-116.063309	22 May - 20 August
Whitewater Delta Trap #2 (WW2)	33.568267	-116.106378	21 May - 19 July
Whitewater Delta Trap #3 (WW3)	33.53837	-116.06584	22 May - 8 July
Whitewater Delta Trap #4 (WW4)	33.53151	-116.08763	21 May - 7 August

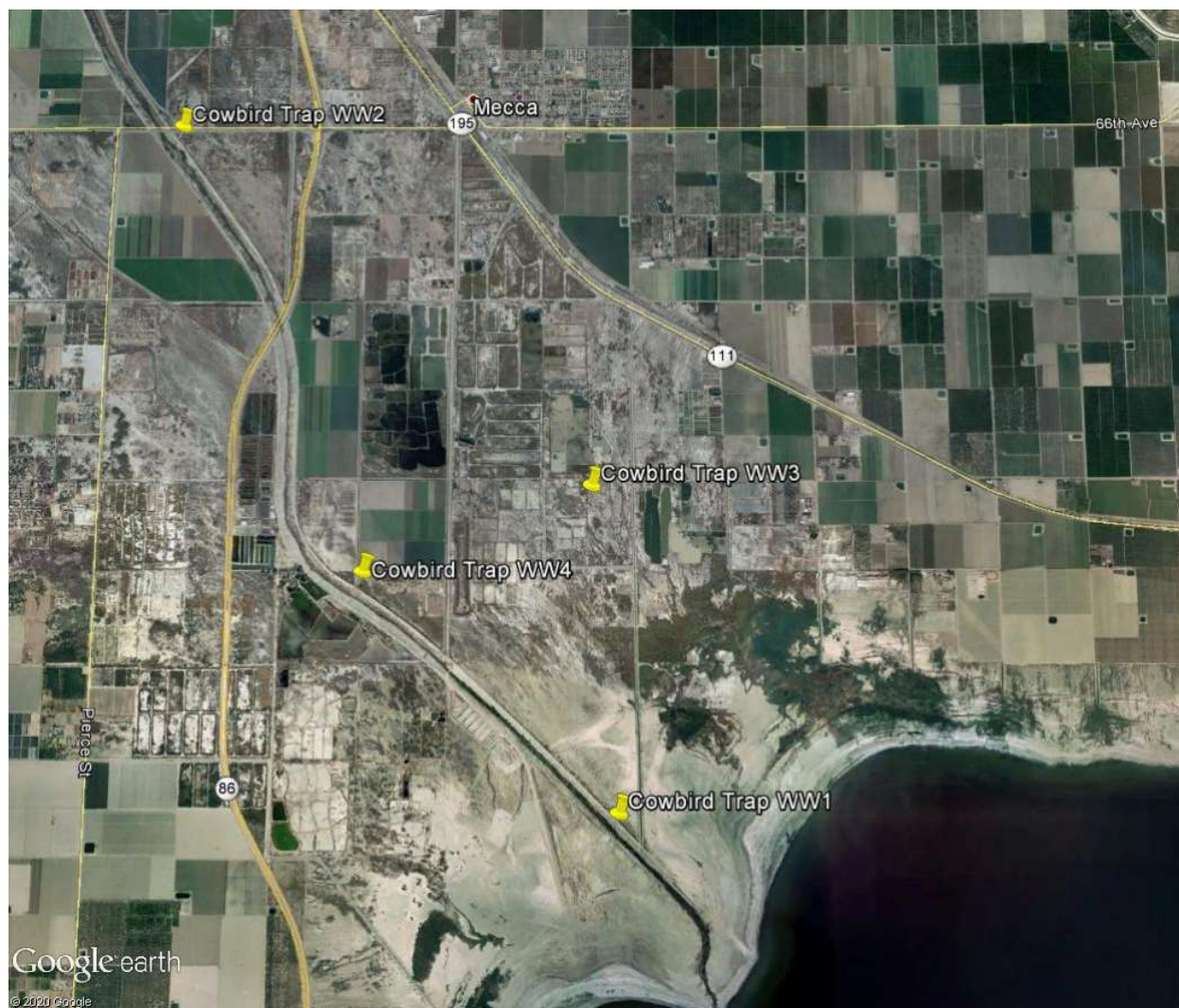


Figure 1. Locations of four cowbird traps in the Coachella Valley in 2020. All four traps were south or west of Mecca, and north of the Salton Sea.



Figure 2. Brown headed Cowbird trap located at site WW1.

Riparian Bird Surveys

Scouting level riparian bird surveys were conducted multiple time between May and August, from the wastewater treatment plant in Indio to the confluence with the Salton Sea (Figure 3, Table 3). The levee roads parallel to the channel were driven, generally during the morning hours, with periodic stops every 200-400 meters to assess the habitat, record the bird species present, and in some cases elicit detections of sensitive bird species through call playback. Bird species detected along the Whitewater Channel are listed in Appendix A.

Table 3. Survey dates and weather conditions for riparian bird surveys at Coachella Valley Stormwater Channel, 2020.

Date	Time	Weather
21 May	1230 - 1800	start: clear, calm, 92°F; end: clear, calm, 90°
22 May	0615 - 1230	start: clear, calm, 70°F; end: clear, S. breeze 5 mph, 90°

16 June	0730 - 1200	start: clear, N. breeze 5 mph, 76°F; end: clear, calm, 96°
17 June	0530 - 1130	start: clear, N. breeze 5 mph, 75°F; end: clear, calm, 90°
7 July	0700 - 1200	start: clear, calm, 80°F; end: clear, calm, 105°F
8 July	0545 - 1130	start: clear, calm, 65°F; end: clear, S. breeze 3 mph, 99°
22 July	0700 - 1130	start: clear, calm, 85°F; end: clear, calm, 96°F
23 July	0630 - 1200	start: clear, calm, 77°F; end: clear, S. breeze 5 mph, 100°F
20 Aug	0800 - 1130	start: clear, calm, humid, 92°F; end: clear, calm, humid, 100°F

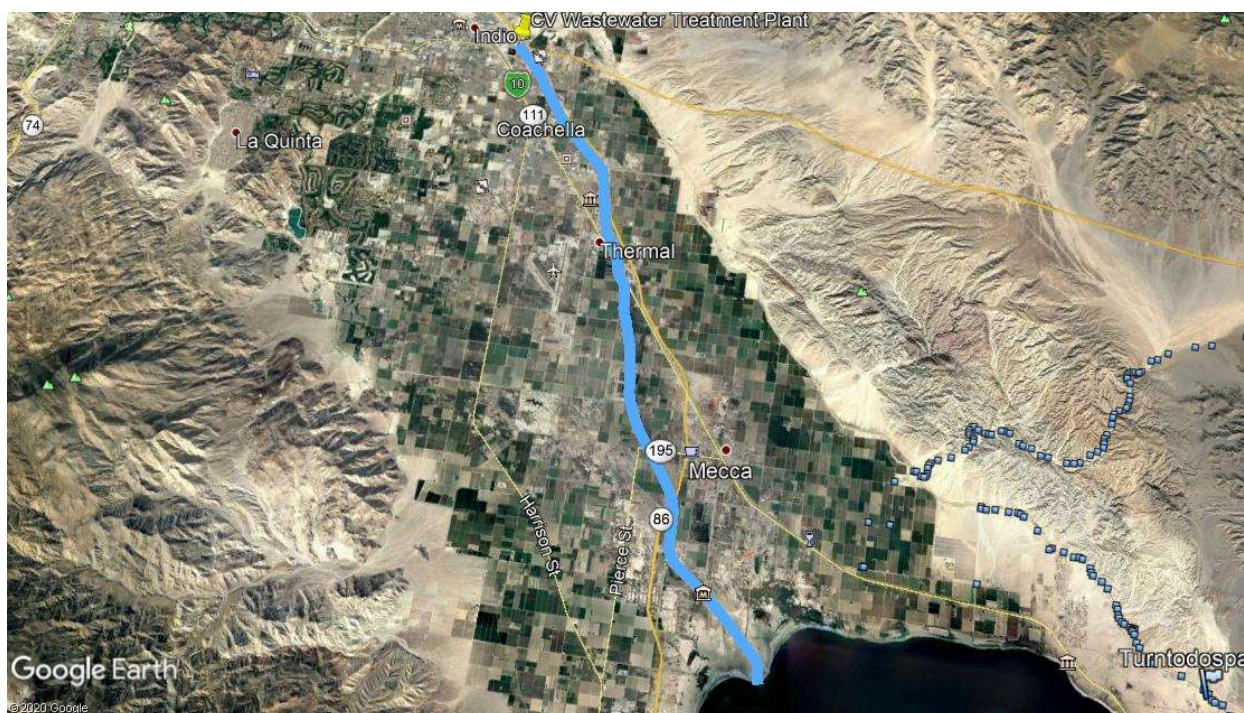


Figure 3. From the wastewater treatment plant in Indio, the Whitewater Channel is a continuously flowing riparian area that continues for 18 miles down to the delta at the Salton Sea.

Results

Cowbird Trapping:

A total of 57 cowbirds were trapped at the four Whitewater Channel traps (Table 4). This includes 18 males, three females, and 36 juveniles. In 2019 a total of 79 cowbirds were trapped including 45 males, 10 females, and 24 juveniles (Figure 4). The much lower ratio of adult females to fledglings trapped in 2020 is likely due to the late start of the trapping, past the peak of the female laying season. In 2019, traps were opened 29-30 April, one month earlier.

The first fledgling cowbird was captured on 31 May, at trap WW1, just nine days after the trap was opened. It was growing in adult feathers, which typically occurs several weeks after fledging, which means this bird probably started as an egg in a parasitized nest in early to mid-April.

Table 4. Summary of cowbird trapping data, Whitewater Channel, 2020. Cowbird numbers do not include recaptures.

Totals	Males	Females	Juveniles	Totals	Bycatch	Dates
WW 1						22 May – 20 August
collected	1	1	11			
Banded/released	4	0	2		HOOR; BUOR; 2 LOSH; ABTO	
Trap Total	5	1	13	19		
WW 2						21 May - 19 July
collected			2			
Banded/released	1	0	0		2 LOSH	
Trap Total	1		2	3		
WW 3						22 May – 8 July
collected			1		ABTO	
Banded/released	0	0	0		GAQU; 2 ABTO	
Trap Total			1	1		
WW 4						21 May -7 August
Collected	10	2	18		ABTO	
Banded/released	2	0	2		4 ABTO; NOMO; 2 LOSH	
Trap Total	12	2	20	34		
All Traps Total	18	3	36	57		

Non-target species captured and released included Abert's Towhee (*Melospiza aberti*; ABTO), Gambel's Quail (*Callipepla gambelii*; GAQU), Loggerhead Shrike (*Lanius ludovicianus*; LOSH), Bullock's Oriole (*Icterus bullockii*; BUOR), Hooded Oriole (*Icterus cucullatus*; HOOR), and Northern Mockingbird (*Mimus polyglottos*; NOMO).

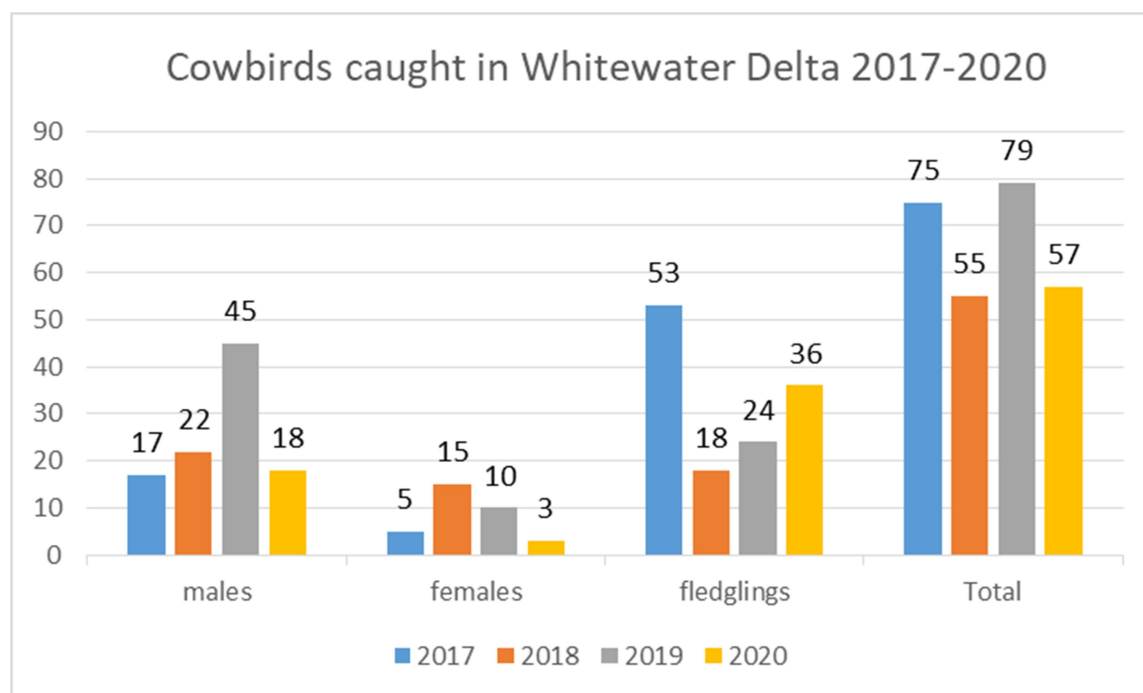


Figure 4. Brown-headed Cowbird captures by year.

Two Abert's Towhees (*Melospiza aberti*) were found dead in the traps, one in WW3 and one in WW4. Each was found in the company of a live Abert's Towhee. These highly territorial birds often attack intruders into their territories, and it is likely that each towhee was killed by the other towhee in the trap. All other non-target birds captured were safely released. Additional non-target species trapped included a single juvenile Gambel's Quail (*Callipepla gambelii*), several adult and juvenile Loggerhead Shrike (*Lanius ludovicianus*), Bullock's Oriole (*Icterus bullockii*), one fledgling Hooded Oriole (*Icterus cucullatus*), and a fledgling Northern Mockingbird (*Mimus polyglottos*; Table 4).

Recaptured Banded Cowbirds:

Five male Brown-headed cowbirds that had been banded in 2019 were captured again in the 2020 season (Table 5). All but one of these had originally been banded and released from a trap in a different location in 2019. Since 2017, 74 male cowbirds have been banded in the area, including eight this year (Table 6), and ten have been recaptured at least one year later. Nine of these have been recaptured within a few miles of their banding location, however one cowbird banded at Dos Palmas Preserve in 2017 was recaptured at the Whitewater Delta in 2018, a distance of 14 miles. These results demonstrate the wide ranging nature of male cowbirds in the area, and that the entire Whitewater Delta area should be considered a single population in terms of management.

Table 5. Banded male cowbirds recaptured in 2020.

Band number	trap	Release Date	Notes
2891-14002	WW2	6 June	First banded as adult at WW2 on 19 June 2019. First captured 4 June 2020.
2891-14024	WW4	26 June	First banded as adult at WW2 on 15 July

			2019.
2891-14031	WW4	26 June	First banded as adult at WW1 on 23 July 2019.
2891-14015	WW2	23 July	First banded as adult at WW3 on 18 June 2019. First captured 23 June 2020.
2891-14026	WW4	23 July	First banded as adult at WW3 on 15 July 2019. First captured 10 June 2020.

Table 6. Male cowbirds newly banded in 2020.

Band number	trap	Date	Notes
2891-14041	WW4	23 July	Hatch year male.
2891-14042	WW4	23 July	Hatch year male.
2891-14043	WW1	23 July	Hatch year male.
2891-14044	WW1	23 July	Hatch year male.
2891-14045	WW1	20 August	Adult male.
2891-14046	WW1	20 August	Adult male.
2891-14047	WW1	20 August	Adult male.
2891-14048	WW1	20 August	Adult male.

Riparian Bird Surveys

The riparian bird surveys documented territorial Yellow-breasted Chat, Yellow Warbler, and Crissal Thrasher along the Whitewater Channel (Figure 5). Approximately five territorial chats were documented, all south of Lincoln Street (Figures 5-6). This is an area that chats have been regularly documented in the past (Hargrove et al 2014). In past years, a Yellow-breasted Chat has also been detected in the channel just south of 66th Avenue adjacent to the cowbird trap at that location, however no chats were detected at that location this year.

A single territorial Yellow Warbler was consistently detected in the tall riparian vegetation just north of the trap at WW1 near Johnson Street (Figure 5). This bird was heard singing into mid-June, and likely represents a territorial male. It's breeding status and whether it was paired or successfully reproduced was not able to be determined.

A Crissal Thrasher was detected singing from the west side of the riparian forest near Johnson Street (Figure 5). A pair of Crissal Thrashers have been detected in this area in the past (Hargrove et al. 2014). In previous years Crissal Thrashers have been detected adjacent to the channel in mesquite woodland near 52nd Avenue (San Diego Natural History Museum 2019b). No thrashers were detected there this year, but the habitat is the same and at least one pair still likely resides here (See further discussion below under Riparian Habitat Conditions).



Figure 5. Locations of sensitive bird species detected in the Whitewater Delta in 2020. The Johnson Street drain enters the channel to the right in the figure. Approximately five territorial Yellow-breasted Chats were detected. Also noted was a territorial Yellow Warbler, and a Crissal Thrasher.



Figure 6. Yellow-breasted Chat singing in the Whitewater Delta area, 7 July, 2020.

On 23 July, 2020, at approximately 11:15 AM, a Yellow-billed Cuckoo was detected in the Whitewater River, below 66th Avenue. A stringer of cottonwoods occurs at this location in the channel, and after playing a cuckoo vocalization, a cuckoo was seen flying along the line of cottonwoods from north to south. The bird was well observed, and followed for approximately 15 minutes as it perched and periodically flew south through the riparian vegetation. A few photos were taken (Figure 7). The bird never vocalized, and after the first 15 minutes flew east over the levee and into the scrub (Figure 8).

The bird was not detected subsequently, and as it never vocalized it was presumably either a migrant or a local female prospecting for a mate. The cottonwoods at this location are 30-40 feet tall and the vegetation is fairly dense (Figure 9), however this patch of mature vegetation covers a relatively small area, likely too small for a breeding cuckoo. At this location a territorial Yellow-breasted Chat has been detected in 2018 and 2019, though not in 2020.



Figure 7. Yellow-billed Cuckoo found on 23 July 2020 in the Whitewater Channel south of 66th Avenue.



Figure 8. The Yellow-billed Cuckoo was first detected in the cottonwoods in the Whitewater Channel, where it continued to forage for fifteen minutes. It then flew east over the levee and out into the scrub and tamarisk beyond the levee.



Figure 9. The habitat where the cuckoo was seen includes a stringer of cottonwood and willow 30-40 feet tall.

Riparian Habitat Conditions

The repeat surveys along the Whitewater Channel revealed a highly variable mix of recently cleared habitat, re-growing riparian marsh and scrub, and fairly mature riparian forest. The most mature and extensive cottonwood-willow forest is south of Lincoln Street. However several other areas of mature riparian forest are notable. One is the mature cottonwood-willow habitat that extends south from 66th Avenue down to the Highway 86 bridge. This area supported the Yellow-billed Cuckoo observed this year, and in past years has supported Yellow-breasted Chats. It is also adjacent to the large conservation parcel just to the east.

Another area with a notable concentration of sensitive species is the area centered at 52nd Avenue. Just to the east of the channel, and west of Highway 86, is an approximately 80 acre mature mesquite woodland that extends for approximately 2.5 linear miles. Sensitive species found in this area include Burrowing Owl (*Athene cunicularia*), Crissal Thrasher, and Palm Springs round-tailed ground squirrel (*Xerospermophilus tereticaudus chlorus*; Figure 10). The owl and thrasher likely forage in the channel as

well as the mesquite woodland, and any ability to allow riparian and scrub vegetation to remain in the channel in this area would be beneficial for those species.

Mature riparian habitat also occurs in the channel north of 66th Avenue up to 61st Avenue. Because this area is adjacent to large date farms, it has a diversity of bird species that are able to use both the riparian habitat and the artificial woodland habitat in the date farms. Species found in this stretch include Red-shouldered Hawk (*Buteo lineatus*), Lark Sparrow (*Chondestes grammacus*), Bullock's Oriole, and Blue Grosbeak (*Passerina caerulea*).



Figure 10. Mature mesquite woodland (80 acres, purple), riparian vegetation (blue), and sensitive species found near Avenue 52.

The most extensive and notable habitat occurs south of Lincoln Street down to the Salton Sea (Figure 11). This area supports the highest diversity and abundance of sensitive species, as well as a diversity of habitats including freshwater marsh, cottonwood-willow riparian forest, and diverse scrub habitats. In March 2020, a series of heavy rainstorms flooded the area within the levees and caused a new low-flow channel to form to the west of the existing one (Figure 11). This new channel has become rapidly colonized by wetland vegetation, and already supports willow trees 10-15 feet tall (Figure 12). The channel also supports significant wetland vegetation (Figure 13). The area between the levees south of Lincoln Street and down to the Salton Sea encompasses nearly 500 acres of potential riparian, wetland, and scrub habitats, and if allowed to develop would significantly add to the available habitat for numerous sensitive species. The lack of infrastructure in the area suggests that allowing vegetation to develop within the levees would pose little risk to surrounding areas.



Figure 11. New channel formed by March floods (blue line) south of Lincoln St. Green shaded area encompasses nearly 500 acres of potential riparian habitat between the existing levees.



Figure 12. Young growth of riparian vegetation in the new channel south of Lincoln St. These willow are 10-15 feet tall in the first season of growth.



Figure 13. Aerial view of habitat in the new channel south of Lincoln Street, looking to the southwest. Photo by Marco Combs.

DISCUSSION AND RECOMMENDATIONS

Cowbird Trapping

The cowbird trapping at Whitewater Delta in 2020 removed 46 cowbirds from the population, yet only three breeding females. Given the large number of fledglings caught, it is clear that a number of breeding females were not captured by the traps. It is likely that by late May when our trapping began, the breeding season is already beginning to wind down for cowbirds and they are not as attracted to the breeding behaviour exhibited by the cowbirds in the traps as they are earlier in the season. Beginning trapping on or around April 1 would increase captures to numbers similar to previous years.

As the last survey of breeding birds in this area was accomplished in 2014, it is unclear if the trapping since 2017 is having the desired effect of increased breeding productivity and population sizes of target bird species, while reducing the overall population of cowbirds. Point count surveys would determine whether common cowbird hosts such as Common Yellowthroat and Song Sparrow are increasing, as well as sensitive breeding species including Yellow-breasted Chat, Yellow Warbler, and Crissal Thrasher. Therefore targeted surveys of breeding birds in this area is advisable.

Recommendations

- Initiate cowbird trapping by April 1 in order to cover the entire breeding season and maximize adult female captures.
- Initiate targeted mist-netting of female cowbirds at sites south of Lincoln St. to increase female cowbird captures in this area, which hosts high numbers of sensitive species.
- Repeat point count surveys to compare current cowbird populations to previous levels.

Riparian Birds

The occurrence of three sensitive species, Yellow Warbler, Yellow-breasted Chat, and Crissal Thrasher, in the channel south of Lincoln Street highlights the high quality habitat at this location. All of these species would benefit from the growth and maturation of the habitat within the levees. Numerous studies have shown that increased vegetation density helps birds to hide their nests from Brown-headed Cowbirds and nest predators (Larison et al. 1998, Saunders et al. 2003, Sharp and Kus 2006). Therefore any increase in the extent and density of the vegetation will be beneficial to nest success for these species.

Recommendations

- Repeat breeding bird surveys to compare current riparian bird populations to previous levels.
- End habitat clearance south of Lincoln Street. The lack of infrastructure in this area and wide space between the levees down to the Salton Sea means any damage from future flooding potential is minimal. The large amount of area (500 acres) available for riparian vegetation if a braided channel is allowed to develop is highly significant, and could support many more territories of sensitive bird species.
- Limit habitat clearing in the 1.3 mile section of channel between 66th Avenue and the Highway 86 bridge. This area supports Yellow-billed Cuckoo and Yellow-breasted Chat and is adjacent to a conservation parcel to the east.
- Limit vegetation clearing adjacent to 80 acre mature mesquite woodland north and south of 52nd Avenue to maximize habitat values in this area, and to increase foraging area for sensitive species including Crissal Thrasher and Burrowing Owl.

Habitat Management

CVMSHCP program goals include the conservation of 3,870 acres of the Coachella Valley Stormwater Channel and Delta Conservation Area, the permanent establishment of 44 acres of Sonoran cottonwood-willow riparian forest, and the conservation of at least 781 acres of core habitat for Crissal Thrasher. Updated habitat maps would help identify if the dynamic habitat conditions within the Whitewater Channel are maintaining 44 acres of Sonoran cottonwood-willow riparian forest. Also the conservation of the 500 acre area within the levees south of Lincoln Street would contribute significantly to the 781 acres of core habitat for Crissal Thrasher identified as a program goal.

It is understood that some vegetation clearing is necessary in the Whitewater Channel to maintain the integrity of the levee system and protect adjacent and downstream properties. Given the high habitat value of riparian and wetland vegetation within the channel, and the sensitive species found there, alternative methods of vegetation management should be explored. One idea would be a rotational system, where habitat is removed in patches or strips over several years, allowing sensitive species to continue to inhabit the area and move between habitat patches of various ages. Additionally, mapping habitat conditions to focus clearing in areas of low habitat quality containing high densities of invasive species such as tamarisk (*Tamarix* spp.) would help improve habitat quality over time.

An interesting habitat management pilot study to benefit birds would be the installation of nest boxes within mature riparian habitat in the channel, especially south of Lincoln Street. Ladder-backed Woodpeckers (*Picoides scalaris*), the primary cavity nesting species in the region, are rare in the area, and numerous secondary cavity nesters that rely on woodpecker cavities for nesting are rare or absent due to lack of available nesting cavities. These include Lucy's Warbler (*Oreothlypis luciae*), Brown-crested Flycatcher (*Myiarchus tyrannulus*), Ash-throated Flycatcher (*Myiarchus cinerascens*), American Kestrel (*Falco sparverius*), Western Screech-Owl (*Megascops kennicottii*), and potentially Elf Owl (*Micrathene whitneyi*).

Recommendations

- Update habitat mapping along and adjacent to the length of the Whitewater Channel to compare Sonoran cottonwood-willow acreage to the 44 acres to be conserved under the CVMSHCP.
- Updated habitat mapping will also help delineate higher and lower quality habitat conditions that can guide vegetation clearing into areas of lower quality.
- Conduct pilot studies to explore ideas for alternatives to extensive vegetation clearances within the Whitewater Channel, especially in areas supporting sensitive species or adjacent to conservation areas.

ACKNOWLEDGMENTS

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Appendix A.

Avian species detected at Coachella Valley Stormwater Channel, 21 May – 20 August 2020.	
Common Name	Scientific Name
Northern Shoveler	<i>Spatula clypeata</i>
American White Pelican	<i>Pelecanus erythrorhynchos</i>
Brown Pelican	<i>Pelecanus occidentalis</i>
Snowy egret	<i>Egretta thula</i>
Great Blue Heron	<i>Ardea herodias</i>
Great Egret	<i>Ardea alba</i>
Green Heron	<i>Butorides virescens</i>
Black-crowned Night-Heron	<i>Nycticorax nycticorax</i>
White-faced Ibis	<i>Plegadis chihi</i>
Turkey Vulture	<i>Cathartes aura</i>
Osprey	<i>Pandion haliaetus</i>
Red-shouldered Hawk	<i>Buteo lineatus</i>
Cooper's Hawk	<i>Accipiter cooperii</i>
American Kestrel	<i>Falco sparverius</i>
Gambel's Quail	<i>Callipepla gambelii</i>
Killdeer	<i>Charadrius vociferus</i>
Whimbrel	<i>Numenius phaeopus</i>
Black-necked Stilt	<i>Himantopus mexicanus</i>
Laughing Gull	<i>Leucophaeus atricilla</i>
Caspian Tern	<i>Hydroprogne caspia</i>
Mourning Dove	<i>Zenaida macroura</i>
Eurasian collared-dove	<i>Streptopelia decaocto</i>









Avian species detected at Coachella Valley Stormwater Channel, 21 May – 20 August 2020.	
White-winged Dove	<i>Zenaida asiatica</i>
Greater Roadrunner	<i>Geococcyx californianus</i>
Yellow-billed Cuckoo	<i>Coccyzus americanus</i>
Burrowing Owl	<i>Athene cunicularia</i>
Lesser Nighthawk	<i>Chordeiles acutipennis</i>
Ladder-backed Woodpecker	<i>Picoides scalaris</i>
Ash-throated Flycatcher	<i>Myiarchus cinerascens</i>
Western Kingbird	<i>Tyrannus verticalis</i>
Loggerhead Shrike	<i>Lanius ludovicianus</i>
Common Raven	<i>Corvus corax</i>
Northern Mockingbird	<i>Mimus polyglottos</i>
Crissal Thrasher	<i>Toxostoma crissale</i>
Marsh Wren	<i>Cistothorus palustris</i>
Bewick's Wren	<i>Thryomanes bewickii</i>
Verdin	<i>Auriparus flaviceps</i>
Black-tailed Gnatcatcher	<i>Polioptila melanura</i>
Northern Rough-winged Swallow	<i>Stelgidopteryx serripennis</i>
Barn Swallow	<i>Hirundo rustica</i>
Cliff Swallow	<i>Petrochelidon pyrrhonota</i>
House Finch	<i>Haemorhous mexicanus</i>
Lark Sparrow	<i>Chondestes grammacus</i>
Song Sparrow	<i>Melospiza melodia</i>
Abert's Towhee	<i>Melozona aberti</i>
Yellow Warbler	<i>Setophaga petechia</i>

Avian species detected at Coachella Valley Stormwater Channel, 21 May – 20 August 2020.	
Common Yellowthroat	<i>Geothlypis trichas</i>
Yellow-breasted Chat	<i>Icteria virens</i>
Blue Grosbeak	<i>Passerina caerulea</i>
Hooded Oriole	<i>Icterus cucullatus</i>
Bullock's Oriole	<i>Icterus bullockii</i>
Red-winged Blackbird	<i>Agelaius phoeniceus</i>
Great-tailed Grackle	<i>Quiscalus mexicanus</i>
Brown-headed Cowbird	<i>Molothrus ater</i>

Appendix XII – Coachella Valley tortoise genetics



Refining genetic boundaries for Agassiz's desert tortoise (*Gopherus agassizii*) in the western Sonoran Desert: the influence of the Coachella Valley on gene flow among populations in southern California

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Abstract

Understanding the influence of geographic features on the evolutionary history and population structure of a species can assist wildlife managers in delimiting genetic units (GUs) for conservation and management. Landscape features including mountains, low elevation depressions, and even roads can influence connectivity and gene flow among Agassiz's desert tortoise (*Gopherus agassizii*) populations. Substantial changes in the landscape of the American Southwest occurred during the last six million years (including the formation of the Gulf of California and the lower Colorado River), which shaped the distribution and genetic structuring of tortoise populations. The area northwest of the Gulf of California is occupied by the Salton Trough, including the Coachella Valley at its northern end. Much of this area is below sea level and unsuitable as tortoise habitat, thus forming a potential barrier for gene flow. We assessed genetic relationships among three tortoise populations separated by the Coachella Valley. Two adjacent populations were on the east side of the valley in the foothills of the Cottonwood and Orocopia mountains separated by Interstate 10. The third population, Mesa, was located about 87 km away in the foothills of the San Bernardino Mountains at the far northwestern tip of the valley. The Cottonwood and Orocopia localities showed genetic affiliation with the adjacent Colorado Desert GU immediately to the east, and the Mesa population exhibited affiliation with both the Southern Mojave and Colorado Desert GUs, despite having a greater geographic distance (0.5x–1.5x greater) to the Colorado Desert GU. The genetic affiliation with the Colorado Desert GU suggests that the boundary for that GU needs to be substantially extended to the west to include the desert tortoise populations around the Coachella Valley. Their inclusion in the Colorado Desert GU may benefit these often overlooked populations when recovery actions are considered.

Highlights

- The complex geologic and climatic history of the Desert Southwest region of the United States during the last 6 million years shaped the past and present distribution, diversity, and evolution of plants and animals.
- Gene flow and connectivity among populations of Agassiz's desert tortoises, a conservation-reliant species, are influenced by landscape features including mountains, low-elevation depressions, and even roads.
- Sub-sea level areas northwest of the Gulf of California occupied by the Salton Trough and the Coachella Valley are unsuitable as tortoise habitat, thus forming a potential barrier for gene flow.
- Comparison of tortoise populations separated by the northern Coachella Valley revealed genetic linkages with the Colorado Desert (part of the Sonoran Desert) genetic unit, despite the presence of the low elevation barrier formed by the valley.
- Genetic connectivity is postulated to be through the mountains at the north end of the valley, and the Colorado Desert genetic unit should be extended to the west to reflect this connectivity.

Keywords:

Agassiz's desert tortoise, Barrier, Conservation, Genetic Boundaries, Genetic Unit, *Gopherus* biogeography, Gulf of California, Salton Trough

Introduction

The Desert Southwest region of the United States has a complex geologic and climatic history that shaped the past and present distribution, diversity, and evolution of plants and animals. Biological patterns in the region are a reflection of the effects of processes driven by tectonic, volcanic, glacial-interglacial cycling, and shorter-term climatic events across widely varying time scales (Dolby et al. 2015). Situated at the tectonic and zoologically important interface of the North American and Pacific Plates (Atwater 1970, Gottscho 2016), the Desert Southwest has been profoundly shaped and influenced by two major geological processes in the last 6 million years: the formation of the Gulf of California (Dolby et al. 2015) and the evolution of the lower Colorado River (Howard et al. 2019), both of which presented aquatic barriers to some plants and animals. Disentangling the effects of those and other geological processes on the evolution of the biota of the region is an ongoing topic of interest to researchers (Dolby et al. 2015, 2019).

Vicariant events in the region caused by these and other geographic barriers to dispersal (e.g., high mountains and deep valleys) play important roles in the genetic structuring of species. Genetic analyses that assess gene flow, or the lack thereof, can serve to identify barriers to dispersal. Syntheses of geographic and genetic analyses can objectively identify critical management units for conservation, defined as genetic units (GUs; Wayne and Morin 2004). As such, genetic analyses are essential for the effective management of species, especially those that are threatened or endangered.

Agassiz's desert tortoise (*Gopherus agassizii* Cooper) is a threatened species with a wide range involving large portions of both the Mojave and Sonoran deserts of Arizona, California, Nevada, and Utah (Luckenbach 1982, Germano et al. 1994). Desert tortoises are obligate herbivores that avoid some of the harshest conditions of their arid environment by constructing deep burrows and spending the majority of their annual cycle sheltering within. With delayed maturity, a long lifespan, and relatively low egg production, they are very sensitive to additive mortality from human activities and subsidized predators (see reviews in Ernst and Lovich 2009, Berry and Murphy 2019). Despite their wide distribution, declines in the densities of tortoises led to the species being listed as threatened under the U.S. Endangered Species Act in 1990. Declines have continued since then (Allison and Mcluckie 2018), despite significant efforts to recover populations (Averill-Murray et al. 2012).

An important part of the recovery plan for the desert tortoise, developed by the U.S. Fish and Wildlife Service (USFWS 2011), is maintaining and managing for genetic variability and linkages among tortoise populations. Linkages are necessary to offset the negative effects of human-caused barriers in the desert, including roads (Latch et al. 2011) and utility-scale wind and solar energy developments (Lovich and Ennen 2011, 2013). Genetic variability of populations of *G. agassizii* is largely a reflection of the

effects of natural geographic barriers on movements (Britten et al. 1997, Murphy et al. 2007, Gaillard et al. 2017). For *G. agassizii*, significant impediments to dispersal and habitation include high mountain ranges and extremely low elevation areas with high summer temperatures (Hagerty et al. 2011), both of which are inhospitable to tortoises. In the Mojave Desert, the latter includes Eureka, Saline, and Death valleys. Similarly, tortoises in the Sonoran Desert of California (including the Colorado Desert subdivision) are all but absent from the low elevation Salton Trough, including the Coachella Valley (Dimmitt 1977, Luckenbach 1982, Berry and Murphy 2019), Cadiz valley, and the lower Colorado River region (Morafka and Berry 2002). Agassiz's desert tortoises tend to prefer valley bottoms and bajadas at moderate elevations (e.g., 300–920 m; Luckenbach 1982) as habitat. Thus, mountain ranges over about 2,000 m, like the New York (2,296 m) and Providence (2,183 m) mountains, are barriers to gene flow (Hagerty and Tracy 2010). At the local scale, variables such as slope and roads may influence tortoise movements and, thus, gene flow (Latch et al. 2011). Not surprisingly, geographic distance also influences gene flow (Murphy et al. 2007, Hagerty and Tracy 2010, Averill-Murray and Hagerty 2014).

Although GUs have been defined for *G. agassizii* (Murphy et al. 2007, Sánchez-Ramírez et al. 2018), these studies did not include sampling from near the southwestern limits of the species range (e.g., populations surrounding the Coachella Valley; Berry and Murphy 2019). The low density or absence of tortoises in much of the area surrounding the Coachella Valley is due to the lack of suitable habitat in such low elevation areas (much of which is below sea level) and environmental conditions that are physiologically challenging for sustaining *G. agassizii* populations (reviewed by Ernst and Lovich 2009, Berry and Murphy 2019). The valley is one of the driest and hottest parts of the Sonoran Desert (Barrows et al. 2010). Its short flowering season for winter annual plants challenges survival of the herbivorous tortoise (Morafka and Berry 2002). Further, large areas of the Coachella Valley are affected by urban development and agriculture (Beatley 1992), rendering it even less suitable for desert tortoises and other wildlife.

Because conditions in the Coachella Valley are not suitable for tortoises, we hypothesize that it acts a barrier to gene flow for disjunct populations of tortoises that occur on the upland slopes of various mountains and bajadas surrounding the valley, including the Chocolate, Orocopa, Cottonwood, Little San Bernardino, San Bernardino, San Jacinto, and Santa Rosa mountains (e.g., Lovich et al. 2015, 2018a; Berry and Murphy 2019). Comparing the genetic composition of three tortoise populations separated by the Coachella Valley allows a preliminary test of our hypothesis. It is important to note that inferring the natural distribution of tortoises today is complicated by intentional introductions by humans (Murphy et al. 2007, Edwards et al. 2010, Edwards and Berry 2013). For example, tortoises are known to have been released in Anza Borrego Desert State Park in the uplands on

the west side of the Salton Trough between 1971 and 1972 (Luckenbach 1982; J. Manning, Washington State University, personal communication), far from our study sites. In addition, possible releases by Native American tribes that used tortoises for food or other cultural purposes (Schneider and Everson 1989) likely occurred throughout the range of the species. The success of those and other introductions depends on the ability of translocated tortoises to survive under local conditions, something that is not assured (Germano and Bishop 2008) and that varies among individuals (Germano et al. 2017).

Herein, we provide the first genetic assessment of tortoise populations surrounding the Coachella Valley. The first two sampled populations were east of the southern portion of the valley and a third was located about 85 km away at the northwestern tip of the valley. If the Coachella Valley is a barrier to tortoise dispersal and gene flow, genetic distances between sampled populations should be greater

than the genetic distances these populations have to adjacent tortoise populations outside of the valley to the east or northwest. Analyses integrate data from 24 short tandem repeats (STRs; microsatellites) from these samples with homologous information from Murphy et al. (2007).

Methods and Materials

Study area

The Salton Trough is the northern extension of the basin occupied by the Gulf of California. It is a pull-apart basin caused by subsidence from oblique extension across strike-slip faults (Brothers et al. 2009). Extending for about 225 km, the Salton Trough in California includes (from north to south) the Coachella Valley, Salton Sea, Imperial Valley, and in Mexico the Colorado River Delta (Fig. 1). The terrestrial portion of the trough extends northwest from the Gulf of California to the San Geronio Pass, near Palm Springs,

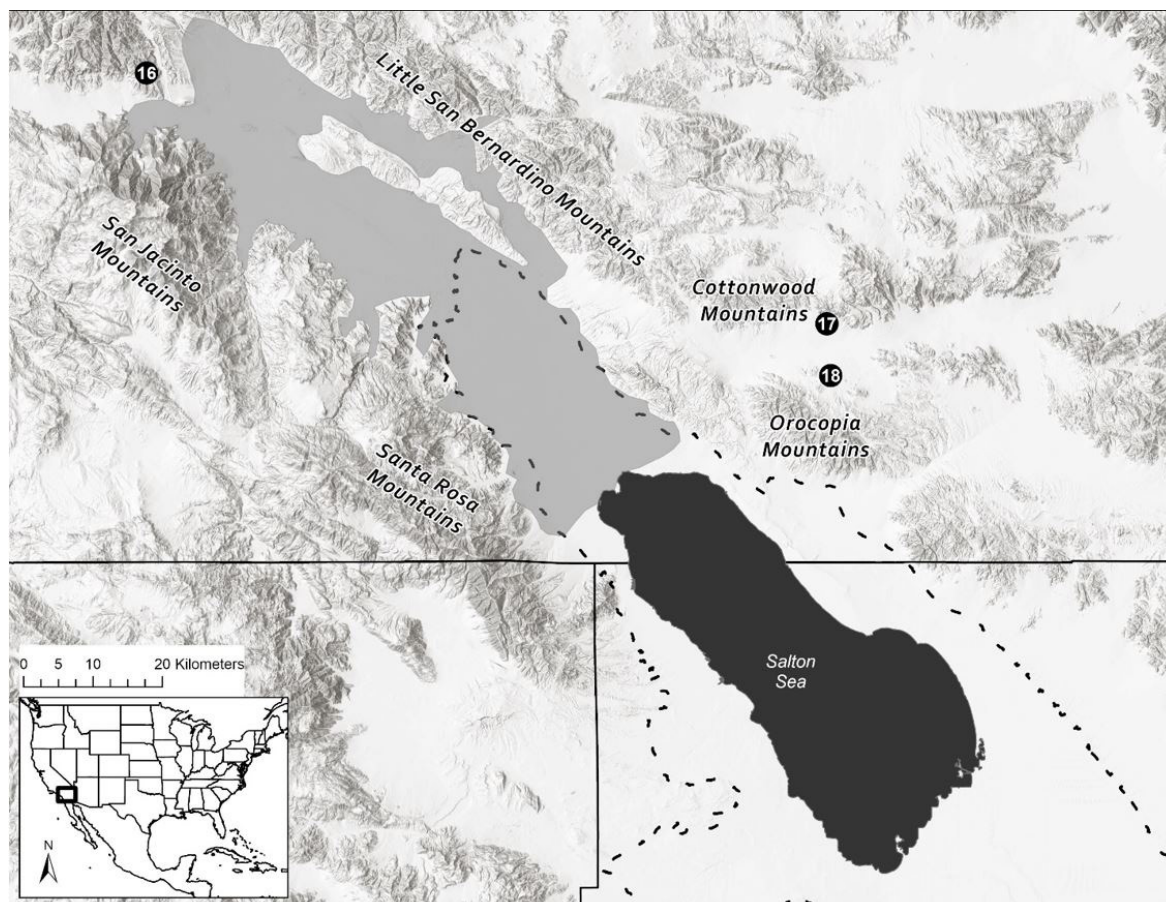


Figure 1. Map showing Agassiz's desert tortoise sampling sites around the Coachella Valley (shaded in gray) in California, USA, the northern extension of the Salton Trough. Study sites are numbered for consistency with other figures and tables: 16=Mesa, 17=Cottonwood, 18=Orocochia. The northwestern part of the Coachella Valley between the label for study site 16 and the San Jacinto Mountains is the San Geronio Pass. The light-colored area around and south of the Salton Sea is a continuation of the terrestrial portion of the Salton Trough before it ends at the Gulf of California in Mexico. County lines are shown for reference. The horizontal line is the southern boundary of Riverside County and the vertical line separates San Diego (left) from Imperial counties (right). Dashed line demarcates the approximate high stand of Lake Cahuilla at approximately 12 m ASL during the Pleistocene obtained from <https://www.arcgis.com/home/item.html?id=ac2b6de1149047b9af934acd4d01fdca>

California. At its lowest point in the Salton Sink, the depression is about 85 m below sea level, although some of that area is now inundated by the Salton Sea with a surface elevation of about 71 meters below sea level. Approximately 5,400 km² of the trough (including the Eastern Coachella Valley) is below sea level, and it is surrounded by mountains on all sides except in the south near the Gulf of California (Waters 1983). The Coachella Valley covers about 777 km² in the northernmost portion of the Salton Trough (Beatley 1992). As mentioned earlier, the low elevation habitats of the Salton Trough, including the Coachella Valley, are unsuitable as tortoise habitat (Nussear et al. 2009) and expected to be barriers to dispersal and gene flow.

The Mesa study site, near Palm Springs, California, is located in the foothills (600–900 m) of the San Bernardino Mountains just above the northwestern tip of the Coachella Valley, on the extreme western edge of the Sonoran Desert ecosystem. The site has been extensively developed for wind energy production (Lovich et al. 2011). Vegetation at the site is a combination of coastal, montane, and desert plant species in a fire-prone landscape (Lovich et al. 2018b). Additional details of the study site and a history of research conducted there are presented by Lovich and Daniels (2000), Agha et al. (2015), and Lovich and Ennen (2017).

The other two study sites (both near Chiriaco Summit, California) are located in the uplands about 28 km due east of the edge of the Coachella Valley floor and about 85 km southeast of Mesa. The Cottonwood study site is located in the southern part of Joshua Tree National Park along the base of the Cottonwood Mountains. The site is characterized by sloping bajadas and desert washes originating in the steep foothills (520–780 m) of the Cottonwood Mountains to the north. Further details of the vegetation in the area are provided by Lovich et al. (2018a). The Orocopia study site is relatively flat, with elevations (480–620m) increasing to the south and east, and a gentle uphill grade to the Orocopia Mountains to the south. This site is characterized by creosote scrub vegetation (*Larrea tridentata* Coville) with widely-scattered ocotillos (*Fouquieria splendens* Engelm.) and blue palo verde trees (*Parkinsonia florida* Watson). The surface is characterized by areas of desert pavement (Wood et al. 2005) interspersed with sandy to gravelly soil and numerous tank and jeep track scars from World War II training activities (Prose 1985) associated with nearby Camp Young in the early 1940's.

Field techniques

From 2000 to 2018, we collected DNA samples from Agassiz's desert tortoises at sites that are outside but hydrologically part of the Coachella Valley (Fig. 2).

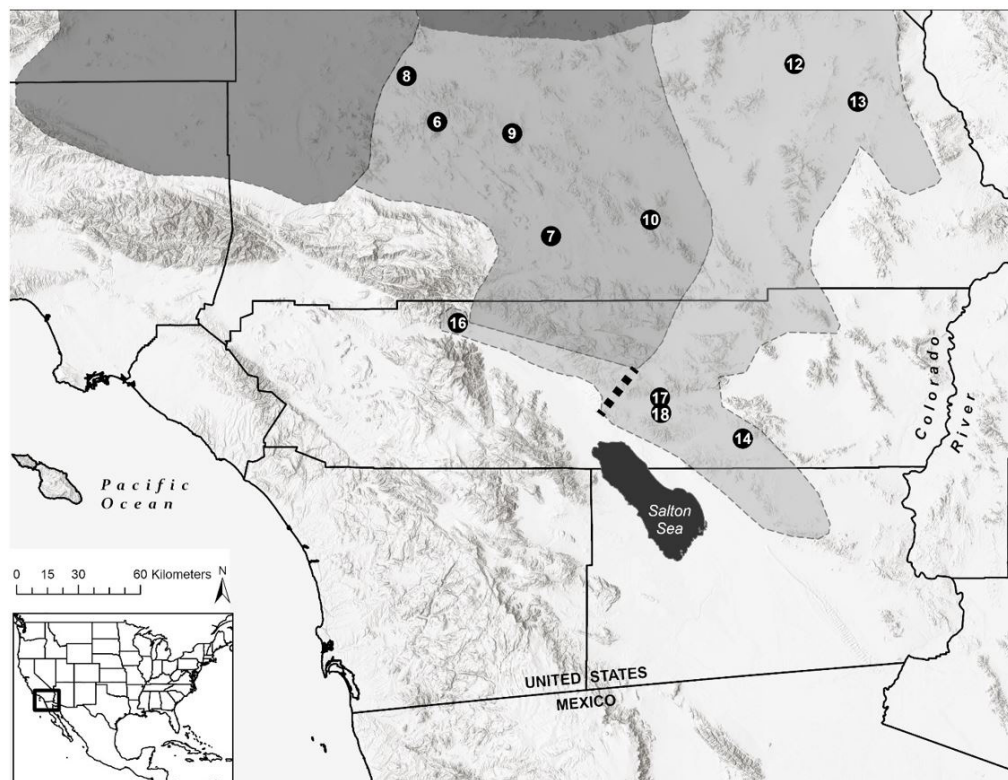


Figure 2. Map showing centroids of locations of groups listed in Table 1. Group 16 is the Mesa study site and Groups 17 and 18 are Cottonwood and Orocopia study sites, respectively. The other sites are listed in Table 1. Shaded areas with fine dashed line borders show the adjacent genetic units modified from Sánchez-Ramírez et al. (2018). From left to right they are: Western Mojave GU, Central Mojave GU (upper sliver), Southern Mojave GU (lower), and Colorado Desert GU. The southern boundary of the Colorado Desert GU is further modified from Sánchez-Ramírez based on our data supporting an extension (left of the bold dashed line) of the GU going northwest from Groups 17 and 18 to Group 16.

We used the subcarapacial venipuncture technique (Hernandez-Divers et al. 2002, Drake et al. 2012) to collect blood and lymph from tortoises, obtaining up to 0.5 ml samples from each adult tortoise using a 23-gauge needle irrigated with sodium heparin. For samples from hatchling tortoises at the Mesa site collected in 2000, we clipped small amounts of keratinous scute tissue from the marginal scutes as part of our individual marking system (Cagle 1939).

Laboratory Processing

A total of 55 samples from our three study sites around the Coachella Valley were processed at the University of Arizona Genetics Core (UAGC), Tucson, AZ, USA. Of these, 31 samples based on scute tissue were received from the University of Southern Mississippi as extracted DNA, and 24 blood samples were extracted and processed at UAGC. Usable DNA was obtained from 53 of the 55 samples.

We mixed whole blood with lysis buffer and incubated overnight with proteinase K at 55°C, followed by robotic extraction using a QIAGEN BioSprint 96 robotic magnetic-particle purification system (Qiagen; Valencia, California, USA) and Aline Biosciences Buccal Swab gDNA Kit (Aline Biosciences; Woburn, Massachusetts, USA). Scute samples were extracted using a Qiagen DNeasy tissue kit. We quantified recovered DNA using a BioTEK Synergy HT (BioTEK; Vermont, USA). We analyzed an approximately 1,100 base pair portion of mitochondrial *ND3/ND4* for samples from Orocopia and Mesa to establish a baseline for the populations and to help identify native vs. translocated individuals (Edwards and Berry 2013). We genotyped all samples for 24 previously described short tandem repeats (STRs) following the methods of Edwards and Berry (2013). Fragment analysis and DNA sequencing were performed following standard procedures by UAGC.

We scored the STR fragment data using Genemarker v.1.85 (SoftGenetics; State College, Pennsylvania, USA). We aligned mtDNA sequences using CLC DNA Workbench v.5.7.1 (CLC Bio; Denmark) to established reference haplotypes in Genbank (Murphy et al. 2007).

Descriptive statistics

We calculated diversity indices for each population based on polymorphic STR loci. We used Arlequin (v. 3.5.1.2; Excoffier and Lischer 2010) to detect significant departures from Hardy-Weinberg expectations and FSTAT v.2.9.3.2 (Goudet 1995) to generate estimates of gene diversity and allelic richness per locus. We used GENEPOP v.4.2 (Raymond 1995, Rousset 2008) to estimate inbreeding coefficients (F_{IS} ; Weir and Cockerham 1984). We used default parameters in FSTAT, ARLEQUIN, and GENEPOP for all Markov-chain tests and permutations.

Population analyses

We compared data from our three study sites ($n=53$ samples) around the Coachella Valley (described above) to a reference database of 709 samples of *G. agassizii* collected from throughout the species' range (Murphy et al. 2007) as updated by Edwards and Berry (2013) for additional loci that match our 24 STR loci. The samples in these previous analyses clustered into 15 groups (our sites 17 and 18 combined into one new group for this analysis) according to Murphy et al. (2007), which we assigned to 5 genetic units (GUs) based on the results of Sánchez-Ramírez et al. (2018) (Table 1). We generated an F_{ST} matrix among all groups using ARLEQUIN. Relationships between genetic and geographic distances were considered qualitatively only because geographic features affected linear travel by tortoises (Dutcher et al. 2020). Thus, relatedness among populations relied on assignment tests using

Table 1. Desert tortoise study sites and groups modified from Murphy et al. 2007, including our three study sites around the Coachella Valley (sites 16, 17, 18). Sites 17 and 18 were combined into one group due to their proximity. All are grouped into 5 genetic units for *G. agassizii* as described in Sánchez-Ramírez et al. (2018). Modified sample set for Southern Clade California samples excludes Northern Clade groups 11, 12 and 15. Abbreviations are as follows: NTC = National Training Center, MCAGCC = Marine Corp Air Ground Combat Center, CMAGR = Chocolate Mountain Aerial Gunnery Range.

Genetic Unit	Study Site	# of Samples	Group	# of Samples in Group
Western Mojave	Desert Tortoise Natural Area	56	1	60
	Fremont-Valley	4	1	
	Hinkley	12	2	83
	Kramer	3	2	
	Edwards Air Force Base	57	2	
	Fremont-Kramer	11	2	
Central Mojave	Superior-Cronese	10	3	52
	Fort Irwin (Goldstone)	9	3	
	Fort Irwin (NTC)	33	3	
	Fort Irwin (Tiefort)	31	4	31
	Fort Irwin (Control Site)	33	5	46
	Fort Irwin (Eastgate 2)	13	5	

Table 1. Continued...

Genetic Unit	Study Site	# of Samples	Group	# of Samples in Group
Southern Mojave	Lucerne Valley	11	6	25
	Ord-Rodman	14	6	
	MCAGCC (Emerson)	9	7	70
	MCAGCC (Sand Hill)	61	7	
	Daggett	72	8	72
	MCAGCC (Lavic Lake)	8	9	27
	MCAGCC (Maumee Mine)	7	9	
	MCAGCC (Sunshine Park)	12	9	
	MCAGCC (Bullion)	16	10	19
	MCAGCC (Lava)	3	10	
Colorado Desert	Fenner	4	12	31
	Goffs	27	12	
	Chemhuevi	7	13	17
	Upper Ward Valley	10	13	
	Chuckwalla	17	14	36
	CMAGR	19	14	
Northern Mojave	Ivanpah	33	11	59
	Ivanpah (site 14)	23	11	
	Shadow Valley	3	11	
	Upper Virgin River. UT	28	15	28
<i>unassigned</i>	Mesa	30	16	30
(Coachella Valley)	Cottonwood	10	17	23
	Orocopia	13	18	

WHICHRUN (Ver. 4.1; Banks and Eichert 2000), which calculates the likelihood of a given individual originating from ≥ 2 candidate populations on the basis of its multilocus STR genotype. We assessed stringency of population allocation based on the log of the odds ratio (LOD) for the two most likely source populations. Assignments with a LOD ratio of ≥ 2 had a ≤ 0.01 chance of type I error.

We used STRUCTURE v.2.3.4 (Pritchard et al. 2000) to assess associations of the Coachella Valley samples with the database without prior population assignments. Analyses were run for $K = 1-12$ with 10 trials per K , and each run for 500,000 iterations following a burn-in period of 50,000 MCMC iterations under the admixture model, assuming allele frequencies were correlated between populations. We used STRUCTURE HARVESTER Online (Earl and von Holdt 2012) to evaluate the results. Independent STRUCTURE runs were compiled and visualized using the Greedy K algorithm in CLUMPAK (Kopelman et al. 2015). We performed analyses two ways: 1) using all samples shown in Table 1, and 2) just the “Southern Clade”. The latter included all locations with mitochondrial “haplogroup” MOJ_A, found throughout most of

California by Murphy et al. (2007), but excluded haplogroup MOJ_B (groups 11, 12, 15 in Table 1) from the northeastern Mojave Desert.

Results

For the three study sites around the Coachella Valley, 34 individuals were sequenced for mtDNA, including 9 from Orocopia and 25 from Mesa; all individuals had haplotype MOJ_A01 (identified by Murphy et al. 2007 as a widely distributed haplotype of haplogroup MOJ_A in California). STRs genotypes were obtained for 53 individuals, including 30 from Mesa, 10 from Cottonwood, and 13 from Orocopia. All populations exhibited heterozygosity consistent with each other (Table 2) and other reported populations of *G. agassizii* (Edwards and Harrison 2014). Allelic richness and gene diversity were relatively low, but this was not unexpected for small populations that are relatively isolated from a geographic perspective (Table 2).

Data for populations at the nearby study sites Cottonwood and Orocopia were combined, and they were most similar (Table 3) to their closest Colorado Desert GU (Group 14; Chuckwalla and CMAGR) in our comparison with the reference database. The Mesa

population was equidistant in relatedness between locations in the Southern Mojave and Colorado Desert GUs, despite having a greater geographic distance (0.5x–1.5x greater) to the former locations. Assignment tests associated most (21/23) Cottonwood and Orocopia individuals to the Colorado Desert GU, whereas individuals from Mesa were split between the Colorado Desert GU and the Southern Mojave GU, and two individuals were assigned further west to the Western Mojave GU (Table 4).

When the Coachella Valley samples were analyzed against the Southern Clade using STRUCTURE

(Figs. 3–4), the best fit was $K = 5$, when evaluated using the DeltaK method of Evanno et al. (2005). In all iterations, Cottonwood, Orocopia, and Mesa samples primarily clustered with the “Colorado Desert” (group 14), with a few exceptions (Table 5). Analysis using all samples in the reference database did not change the clustering. As K was increased, Daggett (group 8, Table 1) fell out as a cluster before any of the unassigned Coachella Valley samples broke away from the Colorado Desert GU (group 14). Mesa became distinct only when evaluated for $K = 7$ (Fig. 3).

Table 2. Mean diversity indices based on 24 microsatellite (STR) loci: n = number of individuals genotyped; richness and diversity estimated using FSTAT. Randomization tests for Hardy–Weinberg equilibrium calculated with ARLQUIN, where Obs Het = observed heterozygosity and Exp Het = expected heterozygosity. F_{IS} , inbreeding coefficient (Weir and Cockerham 1984) estimated using GENEPOP. Italicized values indicate standard deviation of the mean.

Pop	n	# alleles	Allelic range	Allelic richness	Gene diversity	Obs Het	Exp Het	F_{IS}
Mesa	30	6.60 (3.59)	13.85 (9.82)	2.52 (0.61)	0.65 (0.21)	0.655 (0.259)	0.647 (0.221)	0.000
Cottonwood	10	6.28 (2.82)	13.17 (10.59)	2.81 (0.54)	0.74 (0.15)	0.698 (0.216)	0.739 (0.158)	0.059
Orocopia	13	5.80 (3.24)	12.05 (10.14)	2.56 (0.73)	0.65 (0.24)	0.561 (0.274)	0.649 (0.244)	0.147

Table 3. Population pairwise genetic distance (F_{ST} ; below diagonal) and geographic distance (Km; above diagonal) between unassigned populations around the Coachella Valley and adjacent genetic units to the east and north-northwest. Group 16 = Mesa, Group 17 = Cottonwood, and Group 18 = Orocopia. Orocopia and Cottonwood locations combined for analysis due to proximity.

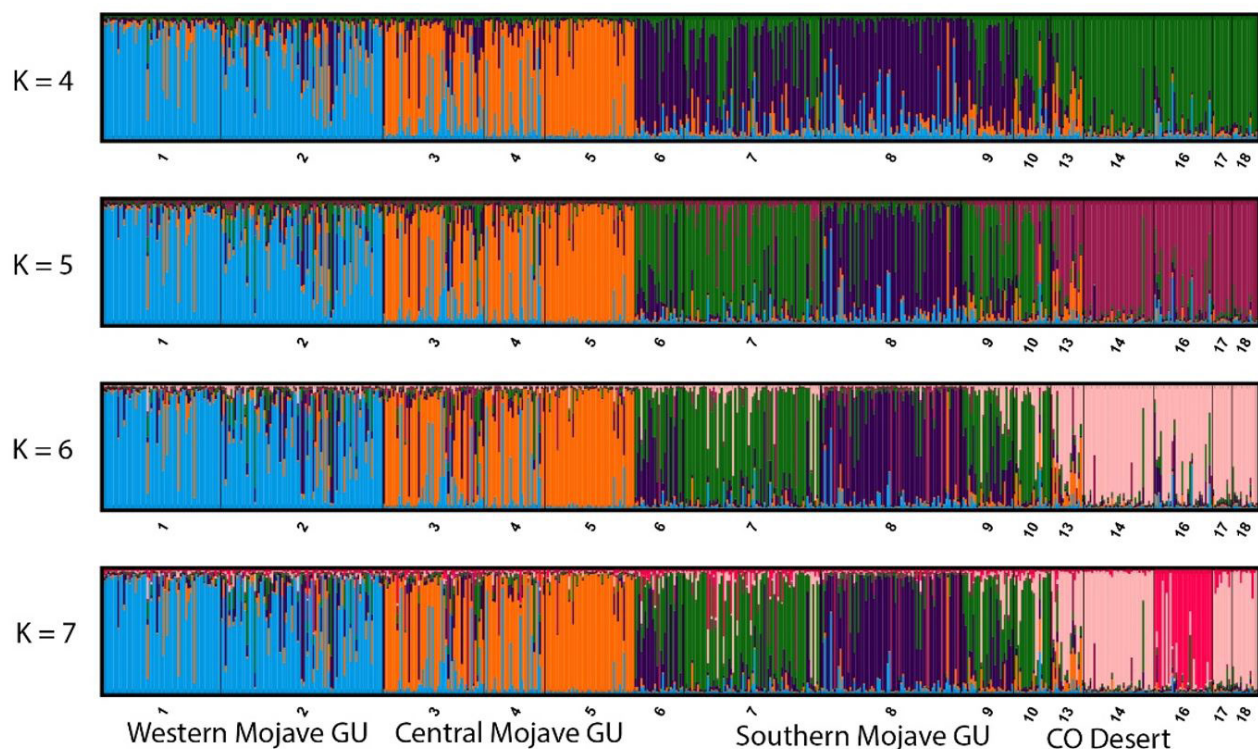
Genetic Unit		Southern Mojave					Colorado Desert	Unassigned (Coachella Valley)	
	Group	6	7	8	9	10	14	16	17&18
Southern Mojave	6	0	49	32	24	77	173	81	144
	7	0.014	0	79	38	35	120	51	81
	8	0.020	0.041	0	52	104	202	100	166
	9	0.014	0.012	0.023	0	55	156	79	124
	10	0.026	0.017	0.047	0.020	0	107	88	74
Colorado Desert	14	0.067	0.047	0.085	0.059	0.045	0	124	36
Unassigned	16	0.056	0.060	0.069	0.060	0.065	0.068	0	87
	17&18	0.068	0.058	0.079	0.060	0.050	0.011	0.054	0

Table 4. Population assignment of three tortoise populations around the Coachella Valley to a reference database of 709 *G. agassizii* samples designated into five genetic units based on the genetic structure reported by Sánchez-Ramírez et al. (2018). Parenthetical values are assignments with $LOD > 2$.

	n	Genetic unit assignment		
		Western Mojave	Southern Mojave	Colorado Desert
Mesa	30	2 (2)	12 (11)	16 (15)
Cottonwood	10	0	1 (1)	9 (9)
Orocopia	13	0	1 (1)	12 (11)

Table 5. STRUCTURE analysis Q-values for K = 5 clusters averaged across 10 iterations. Standard deviations in italics.

Population		Colorado Desert GU	Southern Mojave GU	Southern Mojave GU (Daggett cluster)	Western Mojave GU	Central Mojave GU
Mesa	Ave	0.752	0.100	0.064	0.063	0.021
	SD	<i>0.230</i>	<i>0.137</i>	<i>0.096</i>	<i>0.103</i>	<i>0.026</i>
Orocopia	Ave	0.893	0.042	0.025	0.019	0.019
	SD	<i>0.086</i>	<i>0.057</i>	<i>0.019</i>	<i>0.014</i>	<i>0.019</i>
Cottonwood	Ave	0.899	0.039	0.023	0.015	0.025
	SD	<i>0.049</i>	<i>0.036</i>	<i>0.011</i>	<i>0.007</i>	<i>0.025</i>

**Figure 3.** STRUCTURE analyses for Agassiz's desert tortoise without prior population assignments consolidated and visualized using CLUMPAK for genetic clusters (K) 4–7 for three Coachella Valley populations (16–18) and Southern Clade genetic units; Western Mojave GU (groups 1–2), Central Mojave GU (groups 3–5), Southern Mojave GU (groups 6–10) and Colorado Desert GU (groups 13–14). STRUCTURE analyses histograms consolidated and visualized using CLUMPAK for genetic clusters K = 4–7.

Discussion

Our analyses did not detect deep, lineage divergence among tortoise populations separated by the Coachella Valley, as would be expected if it was a significant barrier to gene flow now or in the past. Instead, the three tortoise populations surrounding the valley show evidence of continuity with the Colorado Desert GU and gene flow consistent with geographic distance, including some evidence of gene flow between Mesa at the northwestern tip of the valley and the adjacent Southern Mojave GU.

The observed lack of differentiation is somewhat unexpected given the long history of geological

change in the region. For example, prehistoric aquatic barriers in the lower Colorado River basin drove divergence between Mojave and Sonoran desert biota (Bell et al. 2010, Dolby et al. 2015). The first barrier, a marine incursion of the Gulf of California, extended northwestward into what is now the Salton Trough as early as 6.3 Ma (Dolby et al. 2019). Fossil-rich marine sediments show that the Gulf extended as far north as San Geronio Pass near Palm Springs, California, and possibly east to Parker, Arizona along what is now the Colorado River (Dolby et al. 2015, 2019). The Colorado River extensional corridor of the Miocene left a series of basins that were flooded in the Pliocene by the precursor to the lower river. Water from the

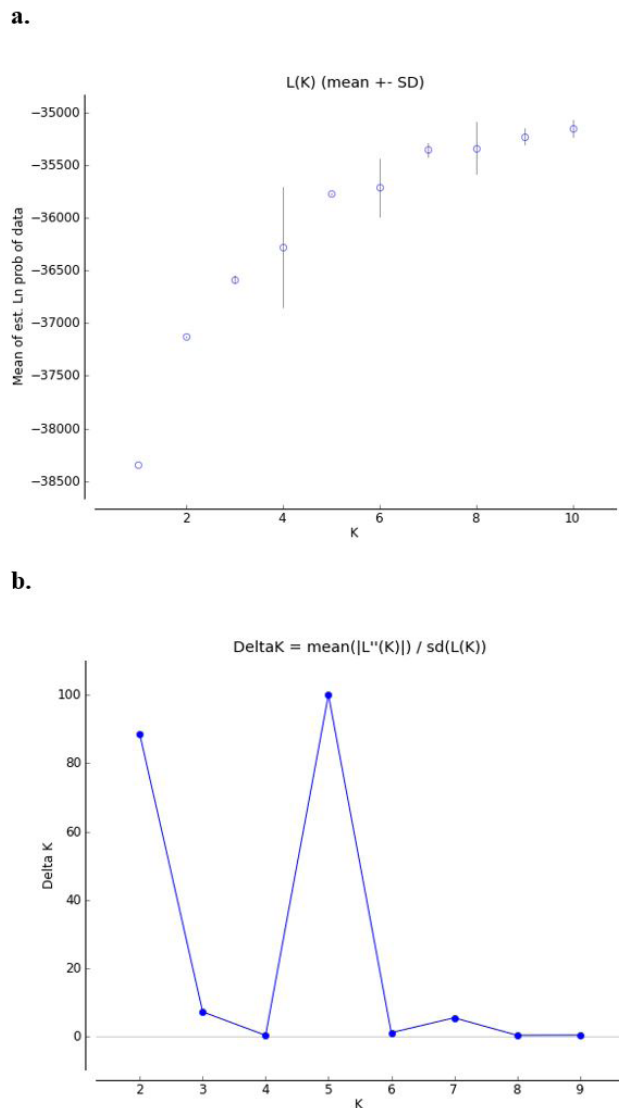


Figure 4. a) The mean log-likelihood of the number of populations of Agassiz's desert tortoise for each K (ranging from 1–10), with error bars representing the standard deviation around the mean. b) DeltaK, derived through the Evanno et al. (2005) method, is shown for each K 1–10.

Colorado River arrived in the lower basin after 5.24 Ma, forming a second barrier (Howard et al. 2019) or filter (Dolby et al. 2019) to dispersal and gene flow in what is now the Salton Trough.

Over the last several thousand years, the Salton Trough has also been inundated periodically with freshwater from overflow of the Colorado River, forming a large waterbody known as Lake Cahuilla (Waters 1983), and it is now partially filled by the hypersaline Salton Sea. Lake Cahuilla (Fig. 1) and its earlier marine precursor would have been dispersal barriers to tortoise movements in the Salton Trough, for while they are capable of floating (Patterson 1973), sometimes for great distances (Gerlach et al. 2006), tortoises are poor swimmers (see review in Strong

and Walde 2006), especially *G. agassizii* (Woodbury and Hardy 1948) since their feet lack webbing present in many aquatic turtles (Pace et al. 2001, Ernst and Lovich 2009).

Aquatic barriers in the Salton Trough and lower Colorado River Valley affected gene flow and evolution not only of tortoises (Lamb et al. 1989), but their impact has also been demonstrated to varying degrees in phylogeographic studies of some other reptile species (e.g., Devitt 2006, Mulcahy et al. 2006), but not others (Lamb et al. 1992, Gottscho et al. 2017).

Past and present conditions in the Salton Trough, including the Coachella Valley, present tortoises with inhospitable habitat that inhibits them from moving from one side to the other in a straight line, but this dispersal barrier does not isolate them. Tortoises may have circumvented the Coachella Valley using its mountainous northern perimeter as a corridor for gene flow. The most likely route of connectivity involves the northern edge of the valley through the foothills, canyons, and bajadas of the Little San Bernardino Mountains. Our analyses cluster together Mesa, Orocopa, and Cottonwood from across the Coachella Valley. Nowadays, Interstate 10 separates Orocopa and Cottonwood, which are only about 6 km apart, and their genetic distance to one another is substantially less than the distance to Mesa. Tortoises at Mesa exhibit evidence of historic gene flow with populations to the north-northwest, yet it still clusters with the Colorado Desert GU.

Our analyses suggest that the range of the Colorado Desert GU should be extended to include populations surrounding the Coachella Valley. Populations around the Coachella Valley appear to show stronger genetic affinities to the Colorado Desert GU to the east (e.g., populations in Chuckwalla and Chocolate Mountain Aerial Gunnery Range, Table 1) than they do to the Southern/Central Mojave Desert GUs to the north. The Little San Bernardino and Cottonwood mountains at the southern boundary of Joshua Tree National Park form a natural barrier for tortoises between the Coachella Valley and the Southern Mojave GU (e.g., Lucerne Valley, Ord-Rodman, MCAGCC, Table 1). The same barrier forms the boundary between the climatically different Mojave and Sonoran desert ecosystems (see Lovich and Bainbridge 1999, Barrows 2011, and references therein), so our results correspond with the zoogeography of the region.

Mesa (Table 1, group 16) appears to have some introgression with the Southern Mojave GU, with decreasing influence moving eastward across the Colorado Desert GU (groups 17, 18 & 14: Fig. 2, Table 5). Thus, historic gene flow at Mesa likely occurred through the Morongo Valley corridor between the San Bernardino and Little San Bernardino mountains, and this is consistent with the model of isolation-by-distance (Murphy et al. 2007, Hagerty et al. 2011) exhibited by *G. agassizii*. No population has been isolated completely over time; gene flow occurs/occurred among neighboring populations.

The shared mtDNA ancestry among the Coachella Valley tortoises suggests that the current distribution

may be attributed either to a recent origin or to unabated dispersal. Edwards (2015) suggested that demographic events or selection reduced diversity within California, leaving tortoises with mtDNA Haplogroup MOJ_A only. This could have involved a population bottleneck followed by a population expansion (Edwards 2003), possibly coinciding with climate change associated with glacial-interglacial periods during the Pleistocene. The estimated time to the most recent common ancestor of this haplogroup—98,268 years (SD $\pm 48,000$ years; Edwards 2003)—is consistent with this hypothesis. Desert tortoises likely experienced multiple population contractions and expansions correlating with Pleistocene glacial and interglacial events. The lower Colorado River Valley appears to have maintained more desert-like conditions during the Wisconsin glacial period (Betancourt et al. 1990), and this area may have acted as a refugium for desert-evolved species. It is unclear how *G. agassizii* may have benefitted from such a refugium since they are not considered to be a desert-evolved species (Morafka and Berry 2002).

The Salton Trough does not appear to be a major driver of population structure in the current distribution of desert tortoises. Instead the Little San Bernardino and Cottonwood mountains appear to limit gene flow to neighboring populations to the north, separating the Southern Mojave GU and the Colorado Desert GU. Although all *G. agassizii* exhibit a fairly recent shared ancestry (within maternal clade MOJ_A), enough time has passed for local adaptation to occur across the species' large distribution (Sánchez-Ramírez et al. 2018). Tortoises around the Coachella Valley likely retain adaptations unique to the lower Colorado River Valley that allow them to persist at the edge of the species' range.

Low-lying desert areas, currently occupied by desert tortoises, are expected to become unsuitable in a warming and drying climate scenario (Barrows 2011, Barrows et al. 2016), and there is some preliminary evidence that the transition is already underway (Lovich et al. 2014). In the context of species conservation, we cannot predict which adaptive traits will be most critical in the face of environmental change and which individuals will contribute most to the evolutionary potential of the species. Individuals living on the edges of their distribution or in marginal habitat have the potential for being better adapted to changing environmental conditions (like climate change) that are different or more challenging from conditions in the core of their distribution (Eckert et al. 2008, Palstra and Ruzzante 2008, Hardie and Hutchings 2010). Desert tortoise populations around the Coachella Valley, especially west of the Salton Trough, represent an underappreciated extension of the species' range. Their genetic affiliation with the Colorado Desert GU and the lower Colorado River Valley, which potentially acted as a refugium during previous climate change events, suggest that it is essential to maintain the connectivity of these populations for the long-term sustainability of the species in that GU.

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