Appendix A
Metadata for Data Layers used in the Draft DRECP Baseline Biology Report

Data Layer	Metadata Description
Aerial imagery	Bing maps. This Microsoft product is updated on a regular basis.
Audubon Important Bird Areas	A product of the Important Bird Areas Program (IBA), Audubon Important Bird Areas are areas identified as vital to birds and other biodiversity that could be targeted for conservation.
BLM Land Designations	Bureau of Land Management land designations, including: Areas of Critical Environmental Concern (ACEC), Special Recreation Management Areas (SRMA), Open Off-Highway Vehicle (OHV) Areas, Source: www.blm.gov/ca/gis/
California Wildlife Habitat Relationships (CWHR) Species Distribution Data September 26, 2008	California Department of Fish and Game's California Wildlife Habitat Relationship System (CWHR) species distribution data. The data is organized into four folders according to the four major taxonomic groups in CWHR: amphibians, reptiles, birds and mammals.
Carbonate Plant Habitat Areas	Mapping of occupied, suitable, and beneficial habitats per the Carbonate Habitat Management Strategy.
Desert Bighorn Sheep Important Areas	Includes the important areas to focus on for conservation of Desert Bighorn Sheep habitat within the Plan Area. Based on data compiled by the California Department of Fish and Wildlife (CDFW) for "A Conservation Plan for Desert Bighorn Sheep in California" and "Optimizing Dispersal and Corridor Models using Landscape Genetics" (Wehausen 2012; Epps et al. 2007). The data consisted of two sets: a raster set showing the mountains with slopes of 15% or greater within the habitat range and a vector set showing the entirety of the intermountain habitat. The intermountain habitat includes low slopes or valley floors with up to 16.4 kilometers between mountain ranges, including stepping stones of mountain habitat between mountain ranges, where applicable. Epps, C.W., J.D. Wehausen, V.C. Bleich, S.G. Torres, and J.S. Brashares. 2007. "Optimizing Dispersal and Corridor Models using Landscape Genetics." *Journal of Applied Ecology* 44(4):714–724. Wehausen, J.D. 2012. "A Conservation Plan for Desert Bighorn Sheep in California." Draft prepared for the California Department of Fish and Wildlife. February 2012.
Desert Linkage Network	Multi-species wildlife corridor modeling from the Desert Linkage Network analysis. A full description of this linkage network development is included in *A Linkage Network for the California Deserts* (Penrod et al. 2012). Penrod, K., P. Beier, E. Garding, and C. Cabañero. 2012. A Linkage Network for the California Deserts. Produced for the Bureau of Land Management and The Wildlands Conservancy. Produced by Science and Collaboration for Connected Wildlands, Fair Oaks, CA www.scwildlands.org and Northern Arizona University, Flagstaff, Arizona http://oak.ucc.nau.edu/pb1/.

Appendix A (Continued)

Data Layer	Metadata Description
Desert Tortoise Priority Areas	Identifies important areas for desert tortoise conservation based on a composite of Tortoise Conservation Areas (USFWS 2011), modeled linkages (Averill-Murray et al. 2013), and habitat potential (Nussear et al. 2009). Averill-Murray, R.C., C.R. Darst, N. Strout, and M. Wong. 2013. "Conserving Population Linkages for the Mojave Desert Tortoise (Gopherus agassizii)." Herpetological Conservation and Biology 8: in press. Nussear, K.E., T.C. Esque, R.D. Inman, L. Gass, K.A. Thomas, C.S.A. Wallace, J.B. Blainey, D.M. Miller, and R.H.Webb. 2009. Modeling habitat of the desert tortoise (Gopherus agassizii) in the Mojave and parts of the Sonoran Deserts of California, Nevada, Utah, and Arizona. U.S. Geological Survey Open-File Report 2009-1102. USFWS (U.S. Fish and Wildlife Service). 2011. Revised Recovery Plan for the Mojave Population of the Desert Tortoise (Gopherus agassizii). Sacramento, California: USFWS.
DRECP Land Ownership Database	The land ownership database is a seamless dataset for the Plan Area used to classify land ownership or public land administration. The dataset was dissolved by landowner, property name, and management type to eliminate multiple polygons for a single property. The dataset was assembled from multiple data sources, including: BLM Land Surface Estate dataset, 2011, http://www.blm.gov/ca/gis/ CDFW-Owned and Operated Lands, 2010 California State Parks Management Boundaries, August 2011 California State Lands Commission ownership dataset, June 2012 GreenInfo Network, 6/2010, 1/2011 – CPAD Database versions 1.5 and 1.6 from http://calands.org
DRECP Landcover Dataset, April 2013	The DRECP Landcover dataset has been assembled from the best available information from multiple sources and has been updated several times during the planning process. The initial land cover map used early in the planning process was a composite dataset created primarily from California Gap (2008 CA-GAP) Vegetation (USGS GAP Program, Lennartz et al. 2008) with updates for agricultural and urban areas from California Farmland Mapping and Monitoring Program (FMMP) (California Department of Conservation 2009). Based on a best-fit strategy (i.e., looking for similarity of species or assemblages), the initial land cover map ecological systems from 2008 CA-GAP were crosswalked to the National Vegetation Classification System (NVCS) "group" level where possible and otherwise to the broader "macrogroup" level. The group level includes combinations of relatively narrow sets of diagnostic plant species, including dominants and co-dominants, broadly similar composition, and diagnostic growth forms. The macrogroup level includes combinations of moderate sets of diagnostic plant species and diagnostic growth forms that reflect biogeographic differences. NatureServe (2009) and Sawyer et al. (2009) vegetation descriptions were used to determine similar community components across vegetation classification systems. Once the land cover map was adapted to the NVCS system, new vegetation mapping

Data Layer	Metadata Description
	conducted in the West Mojave, Lucerne Valley, and East Riverside areas using the NVCS was incorporated into the land cover map using the common classification system (CDFW 2012 and Aerial Information Systems Inc. 2013). Although the new West Mojave mapping data is now mapped and accessible at the alliance level, this finer-scale data is also aggregated to the group level within the common NVCS system to provide a common hierarchical level across the Plan Area for conservation planning purposes. Additionally, datasets from the Joshua Tree National Park and Anza-Borrego Desert State Park were incorporated. The Mojave Desert Ecosystem Project also produced a vegetation map in 2004, which was at mapped at a coarser scale than the alliance level, and this dataset was also incorporated at the group level. The current DRECP land cover map classifies natural communities at the group level across the plan area, and includes a broader "General" level class and a finer-grained alliance level (NVCSName field) class where available. Where alliance level data is not available, the NVCS name repeats the Group level name. In addition to classification attributes, the dataset includes State Rarity ranking and Locally Rare Occurrence designations, as per CDFW 2012. Aerial Information Systems Inc. 2013. *2013 California Vegetation Map in Support of the Desert Renewable Energy Conservation Plan*. Final report. Prepared for California Department of Fish and Wildlife Renewable Energy Program and the California Energy Commission. April 2013. California Department of Conservation. 2009. FMMP dataset. Sacramento, California: Farmland Mapping and Monitoring Program. CDFW (California Department of Fish and Wildlife). 2012. 2012 Vegetation Map in Support of the Desert Renewable Energy Conservation Plan. Interim Report (1.1). Vegetation Classification and Mapping Program for the Desert Renewable Energy Conservation Plan and California Energy Commission. June 2012. https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=47996&inline=1. Lennartz, Steven, Tyler Bax, Jocelyn Aycrigg, Anne Davidson, Marion Reid, and Russ Congalton. 2008. Final Report on Land Cover Mapping Methods. Map Zones 3, 4, 5, 6, 12, and 13. NatureServe. 2009. International Ecological Classification Standard: Terrestrial Ecological Classifications. NatureServe Central Databases. Arlington, VA, U.S.A. Data current as of 06 February 2009. Sawyer, J.O., T. Keeler-Wolf, and J.M. Evens. 2009. A Manual of California Vegetation, Second Edition. California Native Plant Society, Sacramento. 1300 pp. Web Link: A Manual of California Vegetation, Second Edition
DRECP Species Distribution Model Geodatabase	Compiled database of the species distribution models for all Covered Species developed by multiple entities, including CBI, Dudek, UCB, UCD, UCSB, and USGS. Source data and documentation is available on http://databasin.org/.
DRECP Species Occurrence Database, December 2012	Composite database of species localities compiled from multiple sources, including: Audubon and Cornell Lab of Ornithology. eBird Database. May 3, 2011. Audubon golden eagle database (2010, 2011) Bat localities from Pat Brown

Appendix A (Continued)

Data Layer	Metadata Description
	BLM, California Desert District. Point observations of Coachella Valley milkvetch BLM, California Desert District. NECO Occurrence Database (1949-1998) BLM, California Desert District. WEMO 1998 Mohave ground squirrel transect BLM, California Desert District. WEMO Animal, primarily bird, and plant sightings (1968–1996). BLM, California Desert District. WEMO Baseline comprehensive dataset for sightings of animal species with the West Mojave boundary (1956–2001). BLM, California Desert District. WEMO Location of bat roosts within the West Mojave Planning boundary (1978–1998). BLM golden eagle nest location dataset (2012). BLM, Flat-tailed horned lizard Occurrence databases. (2001; 2006; 2007) BLM. Peirson's milk vetch monitoring program (2004-2005) CNDDB, California Natural Diversity Database occurrences, November 2012 CalHERP Arroyo toad occurrences, http://www.californiaherps.com/, April 2012 CDFG. Mojave Ground Squirrel Positive Leitner Points Database. CDFG. Trapping Grid Mojave Ground Squirrel Database. 2005. Leitner. Phil. Leitner Camera Study and Observations; Mohave ground squirrel, 2011-2013. San Bernardino National Forest (SBNF). Spotted Owl Nest Sites. USFWS. Occurrence Information for Multiple Species within Jurisdiction of the Carlsbad Fish and Wildlife Office (CFWO) 2011. USFWS, condor Global Positioning System (GPS) database (2011) USFWS, Peninsular bighorn sheep GPS database, unpublished USFWS; Peirson's milk-vetch database Utah state, flat-tailed horned lizard database Attributing: All of the existing attributes were maintained for each dataset compiled into the DRECP Species Occurrence Database. However, the species scientific name and common name were updated where necessary if they differed from the names listed in Special Animals (CDFG 2011a) or Special Vascular Plants, Bryophytes, and Lichens (CDFG 2011b) in order to maintain consistency with these documents. An attribute for a unique species code was added to each dataset to easily compare the same species across the various sources. Additional attributes were added to reflect currency, validity, and precision to consistently analyze data across the various datasets." Data Currency - Records from before 1990 are coded as "Historic" and records from 1990 to the present are coded as "Current" in the DRECP_Currency field. Records with no date are coded as "unknown" in this field." Validity - All of the records currently included in the database under the DRECP_Validity field are considered valid because each source is data published by a government agency. Additional data that may be added to this database in the future and that does not meet certain criteria for validity could be considered invalid." Precision - The DRECP_Precision field generally follows the precision coding used by the USFWS in their occurrence data. DRECP Precision Codes DRECP Precision Code Definitions USFWS Precision Codes BLM Precision Codes CNDDB Precision Classes 1 within a 160

Data Layer	Metadata Description
	m diameter 1 0-1 specific area; 80 meters 2 within a 500 m diameter 2 - 1/10 mile 3 within a 1 km diameter 3 2-3 1/5 mile 4 within a 2 km diameter 4 4-5 2/5 mile; 3/5 mile 5 within a 4 km diameter 5 6-7 4/5 mile; 1 mile 6 greater than a 4 km diameter 6 8; (blank) 5 miles; D_EXP - CNDDB point data that originated from multi-part polygons that were exploded and a point was forced inside the polygon are flagged with a "YES" value. D_PUBLIC - publically available data flagged with a "YES" value. Multi-part records were "exploded" to yield the actual locations of multiple points associated with single records/element occurrences; therefore, the DRECP species occurrence database, in some cases, has more point locations than the number of element occurrences reported from CNDDB. This was done to enable a fine-scale analysis with greater geographic specificity than would be able otherwise. It increased the accuracy of the intersection of species occurrences with other geographic variables in the Plan Area.
Dunes and Sand Area	Based on a composite of a selection set from the DRECP land cover map that included "North American warm desert dunes and sand flats," a selection set from the surficial geology dataset that included "Sand dunes" (California Department of Conservation 2000), and California desert sand dunes mapping (Dean 1978). California Department of Conservation. 2000. "Geological Map of California." Geographic information system (GIS) data. Dean, Leslie E. 1978. "The California Desert Sand Dunes." Department of Earth Sciences, University of California, Riverside. Jointly Supported by National Aeronautic and Space Administration, Grant No. NSG-7220, and Department of the Interior, Bureau of Land Management. June 1978.
Ecoregion Subsection	The U.S. Forest Service (USFS) (1997) defined ecological sections and subsections (i.e., ecoregions) within California as part of the USFS National Hierarchical Framework adopted by the USFS Ecological Classification and Mapping Task Team (ECOMAP). These ecoregion sections are classified as Level III Ecoregions of the Continental United States by the U.S. Environmental Protection Agency (EPA) (EPA 2003). EPA (Environmental Protection Agency). 2003. "Level III and IV Ecoregions of the Continental United States." EPA – Western Ecology Division. Updated February 13, 2012. Accessed March 1, 2012. http://www.epa.gov/wed/pages/ecoregions/level_iii_iv.htm. USFS. 1997. "Pacific Southwest Region R5-EM-TP-005." In *Ecological Subregions of California: Section and Subsection Descriptions*. Compiled by S.R. Miles and C.B. Goudey. Accessed August 22, 2007. http://www.fs.fed.us/r5/projects/ecoregions/toc.htm.
Elevation Range (Topography), Percent Slope, and Aspect	Elevation range, percent slope, and aspect are derived from the USGS 30 Meter Digital Elevation Model (DEM).
Flat-Tailed Horned Lizard Management Area	Flat-tailed horned lizard Management Areas. Flat-tailed Horned Lizard ICC (Interagency Coordinating Committee). 2003. *Flat-tailed Horned Lizard Rangewide Management Strategy*. 2003 revision. 80 pp. plus appendices.

Appendix A (Continued)

Data Layer	Metadata Description
Hydrology (including Major river, Minor Drainages, Stream/River, and Canal/Ditch)	The National Hydrography Dataset (NHD) is a feature-based database that interconnects and uniquely identifies the stream segments or reaches that make up the nation's surface water drainage system. NHD data was originally developed at 1:100,000-scale and exists at that scale for the whole country. This high-resolution NHD, generally developed at 1:24,000/1:12,000 scale, adds detail to the original 1:100,000-scale NHD. (Data for Alaska, Puerto Rico and the Virgin Islands was developed at high-resolution, not 1:100,000 scale.) Local resolution NHD is being developed where partners and data exist. The NHD contains reach codes for networked features, flow direction, names, and centerline representations for areal water bodies. Reaches are also defined on waterbodies. The NHD also incorporates the National Spatial Data Infrastructure framework criteria established by the Federal Geographic Data Committee. Derived from the NHD PLus Flowlines created by USGS.
Known Geothermal Resource Areas	California Department of Conservation Division of Oil, Gas, and Geothermal Resources data on Known Geothermal Resource Areas (KGRAs).
Land use dataset	Assemblage of county land use information from county sources (San Diego, Imperial, Inyo, Kern, Los Angeles, Riverside, and San Bernardino) and the Southern California Association of Governments.
Landform	Landform is derived from the Land Facet tool using USGS digital elevation model (DEM) data. This data layer classifies areas as ridgelines, plains, valleys, or slopes.
Lane Mountain Milk-Vetch Conservation Area	Mapping of BLM Lane Mountain Milk-Vetch conservation areas for the West Mojave Plan.
Los Angeles County Significant Ecological Areas	Important landscape features in the Los Angeles County region; include washes, Joshua tree woodlands, and important landforms. This is considered a landscape/ecological process element. "Significant Ecological Area" means an area that is determined to possess an example of biotic resources that cumulatively represent biological diversity, for the purposes of protecting biotic diversity, as part of the Los Angeles County General Plan or the city's general plan. Purpose is to identify areas with Significant Ecological Importance, a designation that was adopted with the 1980 General Plan.
Microphyll Woodlands	Based on a selection set from the DRECP land cover dataset (based on CDFG 2012) that included the following: Blue palo verde–ironwood woodland (*Parkinsonia florida–Olneya tesota*), Smoke tree woodland (*Psorothamnus spinosus*), Honey mesquite riparian form (*Prosopis glandulosa*), and Desert willow (*Chilopsis linearis*), as well as the desert wash woodland selection from the vegetation map used in the BLM Northern and Eastern Colorado Coordinated Management Plan (BLM 2002).
Mohave Ground Squirrel Important Areas	Includes the important areas to focus on for conservation of Mohave ground squirrel habitat within the Plan Area. Includes data based on the original Leitner 2008 work and revised in 2012 based on input from Leitner and other Mohave ground squirrel

Data Layer	Metadata Description
	experts. The habitats were defined using field observations; historical and current species occurrence records; habitat suitability, including disturbance analysis and the U.S. Geological Survey 2013 Habitat Suitability Model (Inman et al 2013); expert input; and topography. The following areas were described: population centers, habitat linkages, habitat expansion areas, and climate change extensions. Leitner, P. 2008. "Current Status of the Mohave Ground Squirrel." *Transactions of the Western Section of the Wildlife Society* 44:11–29. Inman, R.D., T.C. Esque, K.E. Nussear, P. Leitner, M. Matocq, P. Weisberg, T. Dilts, and A. Vandergast. 2013. "Is There Room for All of Us? Renewable Energy and *Xerospermophilus mohavensis*." *Endangered Species Research* 20:1–18. doi: 10.3354/esr00487.
Mojave Fringe-Toed Lizard Conservation Areas	Mapping of BLM Mojave Fringe-toed Lizard conservation areas for the West Mojave Plan.
Mojave Monkeyflower Conservation Areas	Mapping of BLM Mojave Monkeyflower conservation areas for the West Mojave Plan.
Mountain ranges	Digitized mountain ranges from DFW
North American Migration Flyways 2012	Migration flyways in the North America, including the Atlantic Flyway, Mississippi Flyway, Central Flyway, and Pacific Flyway from www.birdnature.com.
NWI Wetlands	This data set represents the extent, approximate location and type of wetlands and deep-water habitats in the conterminous United States as defined by the USFWS's National Wetlands Inventory (NWI). These data delineate the areal extent of wetlands and surface waters as defined by Cowardin et al. (1979). Certain wetland habitats are excluded from the National mapping program because of the limitations of aerial imagery as the primary data source used to detect wetlands. These habitats include seagrasses or submerged aquatic vegetation that are found in the intertidal and subtidal zones of estuaries and near shore coastal waters. Some deepwater reef communities (coral or tuberficid worm reefs) have also been excluded from the inventory. These habitats, because of their depth, go undetected by aerial imagery. By policy, the Service also excludes certain types of "farmed wetlands" as may be defined by the Food Security Act or that do not coincide with the Cowardin et al. definition. Contact the Service's Regional Wetland Coordinator for additional information on what types of farmed wetlands are included on wetland maps.
Soil Texture	Soil texture comes from the USDA National Resources Conservation Service (NRCS) Soil Survey Geographic Database (SSURGO). SSURGO Soils Survey - processed for Depth to Any Soil Restrictive Layer, Depth to Water Table, Drainage Class, Ecological Site Name, Hydric Rating, Map Unit Name, Parent Material Name, Soil Taxonomy and Surface Texture.

Appendix A (Continued)

Data Layer	Metadata Description
South Coast Missing Linkages (SCML) Wildlife Corridors	A product from South Coast (SC) Wildlands, an organization working to maintain and restore connections between wildlands in the South Coast Ecoregion. The South Coast Missing Linkages Project addresses fragmentation at a landscape scale by identifying and prioritizing linkages that conserve essential biological and ecological processes. This project gathers the most current biological data for each linkage design to ensure the viability of the full complement of species native to the region.
Springs/Seeps and Wells	Derived from the DRECP NHD Point data, which is a dataset created by USGS and includes hydrologic point features.
Surficial geology/ Soil parent material	California Geology Units from Jennings 1977 Geologic map of California. (California Division of Mines).
TNC Ecoregional Assessment data	Areas identified as moderately degraded and highly converted as defined by The Nature Conservancy (TNC). Marshall, R.M., S. Anderson, M. Batcher, P. Comer, S. Cornelius, R. Cox, A. Gondor, D. Gori, J. Humke, R. Paredes Aguilar, I.E. Parra, S. Schwartz. 2000. *An Ecological Analysis of Conservation Priorities in the Sonoran Desert Ecoregion*. Prepared by The Nature Conservancy Arizona Chapter, Sonoran Institute, and Instituto del Medio Ambiente y el Desarrollo Sustentable del Estado de Sonora, with support from the Department of Defense Legacy Program, and agency and institutional partners. April 2000. Randall, J.M., S.S. Parker, J. Moore, B. Cohen, L. Crane, B. Christian, D. Cameron, J. MacKenzie, K. Klausmeyer, and S. Morrison. 2010. *Mojave Desert Ecoregional Assessment*. Unpublished Report; version 1.1. San Francisco, California: The Nature Conservancy. September 2010. Accessed May 2013. http://conserveonline.org/workspaces/mojave/documents/mojave-desert-ecoregional-2010/@@view.html.
USFWS Designated Critical Habitat	These data identify, in general, the areas where final critical habitat exists for species listed as endangered or threatened. Designated Critical Habitat includes areas considered essential for the conservation of federally listed species. These areas provide notice to the public and land managers of the importance of these areas to the conservation of this species. Special protections and/or restrictions are possible in areas where federal funding, permits, licenses, authorizations, or actions occur or are required.
USFWS. Condor GPS Database. Unpublished. 2011.	Dataset of GPS transmitted data from the USFWS. These data represent a subset of known locations of a subset of California Condors outfitted with GPS tracking devices. Absence of observations do not indicate lack of presence of the species. Furthermore, only a small number of Condors are tracked and untracked birds may be present within the geographic extent represented by these data. The dataset ranges from 2002 to May 9, 2011.

Appendix A (Continued)

Data Layer	Metadata Description
USFWS. Peninsular bighorn sheep GPS Database. Unpublished.	Dataset of GPS transmitted data from the USFWS. This database was established to map known occurrence locations of Peninsular bighorn sheep in conjunction with the Peninsular bighorn sheep Recovery Plan, the critical habitat designation, and Section 7 consultations. It contains known occurrence locations of Peninsular bighorn sheep derived from various sources and covers a range of dates.
USGS topographic maps	1:24,000-scale topographic maps, also known as 7.5 minute quadrangles.
Watershed	The California Interagency Watershed Map of 1999 (updated May 2004, "calw221") is the State of California's working definition of watershed boundaries. Previous Calwater versions (1.2 and 2.2) described California watersheds, beginning with the division of the State's 101 million acres into ten Hydrologic Regions (HR). Each HR is progressively subdivided into six smaller, nested levels: the Hydrologic Unit (HU, major rivers), Hydrologic Area (HA, major tributaries), Hydrologic Sub-Area (HSA), Super Planning Watershed (SPWS), and Planning Watershed (PWS). At the Planning Watershed (the most detailed level), where implemented, polygons range in size from approximately 3,000 to 10,000 acres. At all levels, a total of 7035 polygons represent the State's watersheds. The present version, Calwater 2.2.1, refines the watershed coding structure and documentation (database fields were added and some were renamed). There are significant watershed boundary, code, and name differences between Calwater versions 1.2 (1995), 2.0 (1998), and 2.2 (1999). The differences between versions 2.2 (1999) and 2.2.1 (2004) are attribute field names and some inserted lines that identify differences between State and federal watersheds.

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APPENDIX B

Species Profiles

Appendix B
Species Profiles

Covered Species

There are 37 taxa considered for coverage (i.e., Covered Species) for the Desert Renewable Energy Conservation Plan (DRECP). Table B-1 lists all of the Covered Species. Following Table B-1 are the species profiles for each Covered Species. Species profiles are presented in the order they appear in Table B-1.

Table B-1
Proposed Covered Species List

Taxa	Common Name	Scientific Name	Federal Status[1]	State Status[2]
Amphibian/ Reptile	Agassiz's desert tortoise	*Gopherus agassizii*	FT	ST
	flat-tailed horned lizard	*Phrynosoma mcallii*	BLM/FS	CSC
	Mojave fringe-toed lizard	*Uma scoparia*	BLM	CSC
	Tehachapi slender salamander	*Batrachoseps stebbinsi*	BLM/FS	ST
Bird	Bendire's thrasher	*Toxostoma bendirei*	BCC/BLM	CSC
	burrowing owl	*Athene cunicularia*	BLM	CSC
	California black rail	*Laterallus jamaicensis coturniculus*	BCC/BLM	ST
	California condor	*Gymnogyps californianus*	FE	SE/FP
	Gila woodpecker	*Melanerpes uropygialis*	BLM/BCC	SE
	golden eagle	*Aquila chrysaetos*	BLM	FP
	greater sandhill crane	*Grus canadensis tabida*	BLM/FS	ST/FP
	least Bell's vireo	*Vireo bellii pusillus*	FE/BCC	SE
	mountain plover	*Charadrius montanus*	BCC/BLM	CSC
	Swainson's hawk	*Buteo swainsoni*	BLM/FS	ST
	tricolored blackbird	*Agelaius tricolor*	FC/BCC/BLM	CSC
	western yellow-billed cuckoo	*Coccyzus americanus occidentalis*	FC/FS/BCC/BLM	SE
	willow flycatcher (including southwestern)	*Empidonax traillii (*including *extimus*)	Southwestern: FE	SE
	Yuma clapper rail	*Rallus longirostris yumanensis*	FE/BCC	ST/FP
Fish	desert pupfish	*Cyprinodon macularius*	FE	SE
	Mohave tui chub	*Siphateles (Gila) bicolor mohavensis*	FE	SE/FP

Table B-1
Proposed Covered Species List

Taxa	Common Name	Scientific Name	Federal Status[1]	State Status[2]
	Owens pupfish	*Cyprinodon radiosus*	FE	SE/FP
	Owens tui chub	*Siphateles (Gila) bicolor snyderi*	FE	SE
Mammal	Desert bighorn sheep	*Ovis canadensis nelsoni*	BLM	FP*
	California leaf-nosed bat	*Macrotus californicus*	BLM/FS	CSC
	Mohave ground squirrel	*Xerospermophilus mohavensis*	BLM	ST
	pallid bat	*Antrozous pallidus*	BLM/FS	CSC
	Townsend's big-eared bat	*Corynorhinus townsendii*	BLM/FS	CSC/Candidate
Plant	alkali mariposa-lily	*Calochortus striatus*	BLM	(CRPR 1B.2)
	Bakersfield cactus	*Opuntia basilaris* var. *treleasei*	FE	SE (CRPR 1B.1)
	Barstow woolly sunflower	*Eriophyllum mohavense*	BLM	(CRPR 1B.2)
	desert cymopterus	*Cymopterus deserticola*	BLM	(CRPR 1B.2)
	Little San Bernardino Mountains linanthus	*Linanthus maculatus*	BLM	(CRPR 1B.2)
	Mojave monkeyflower	*Mimulus mohavensis*	BLM	(CRPR 1B.2)
	Mojave tarplant	*Deinandra mohavensis*	BLM	SE (CRPR 1B.3)
	Owens Valley checkerbloom	*Sidalcea covillei*	BLM	SE (CRPR 1B.1)
	Parish's daisy	*Erigeron parishii*	FT	(CRPR 1B.1)
	triple-ribbed milk-vetch	*Astragalus tricarinatus*	FE	(CRPR 1B.2)

Notes:

1 Federal Status - FE: Federally Endangered; FT: Federally Threatened; FC: Federal Candidate Species; FS: Forest Service sensitive; BLM: Bureau Land Management sensitive; BCC: Bird of Conservation Concern

2 State Status - SE: State Endangered; ST: State Threatened; CSC: California Species of Concern; FP: Fully Protected; *: limited hunting; CRPR: California Rare Plant Rank. See https://www.cnps.org/cnps/rareplants/ranking.php for an explanation of CRPRs.

Agassiz's Desert Tortoise
(Mojave Population)
(*Gopherus agassizii*)

Photo by Dudek.

Legal Status

State: Threatened

Federal: Threatened

Critical Habitat: Critical habitat was designated for the Beaver Dam Slope (Utah) population in 1980 (FR 45 55654–55666). Critical habitat for the Mojave population was designated in 1994 (FR 59 5820–5886). See Figure 3 for the location of critical habitat.

Recovery Planning: The original recovery plan for the Mojave population was completed in 1994 (USFWS 1994). A revised draft recovery plan was completed in 2008 (USFWS 2008), and a final revised recovery plan was released in 2011 (USFWS 2011a).

Taxonomy

The generic assignment of the desert tortoise has gone through a series of changes since its original description by Cooper (1863) as *Xerobates agassizii*. Currently, the accepted scientific name is *Gopherus agassizii* (Crumly 1994). Other tortoise species known to be extant in North America, all belonging to the genus *Gopherus*, include Texas tortoise (*G. berlandieri*) that occurs in southern Texas and northeastern Mexico, and the gopher tortoise (*G. polyphemus*) that occurs in southwestern South Carolina, Florida, Georgia, Alabama, Mississippi, Louisiana, and extreme southeastern Texas. The Mexican species is the Bolson tortoise (*G. flavomarginatus*), which occurs in a very small area in Chihuahua and Durango, Mexico (Bury and Germano 1994; USFWS 2011a). Fossils of late Pleistocene *G. agassizii* have been found in the area of McKittrick, California (Miller 1942), with other specimens found as far east as southeastern New Mexico (Moodie and Van Devender 1979).

A recent taxonomic review has formally split the previous single desert tortoise species into two distinct species—Agassiz's (Mojave population) desert tortoise (*Gopherus agassizii*) and Morafka's (Sonoran population) desert tortoise (*G. morafkai*) (Murphy et al. 2011). Agassiz's desert tortoise occurs in southeastern California, southern Nevada, southwestern Utah, and northwestern Arizona. Morafka's desert tortoise occurs in southwestern Arizona and south into Mexico. This genetic study, utilizing mitochondrial DNA, supports long-time observations by desert tortoise biologists that there are distinct differences in ecology, behavior, and life history between tortoises found west and north of the Colorado River, and those found to the south and east.

Although there are genetic and ecological differences between desert tortoises that belong to the Sonoran population, animals attributed to this population could be confused visually with individuals of the Mojave population. Because the visual differences between these populations are minor, the U.S. Fish and Wildlife Service (USFWS) determined at the time of federal listing that the Sonoran population also warranted protection as a threatened species under Section 4(e) of the Endangered Species Act (similarity of appearance) when located outside of its natural range (USFWS 2011a; see also Averill-Murray 2011).The recent taxonomic treatment of the desert tortoise to two distinct species does not affect the listing status of Agassiz's desert tortoise throughout its range.

Distribution

General

The Agassiz's desert tortoise is associated with the Sonoran (Colorado phase) and Mojave Deserts in the southwestern United States (Figure 1). Generally, its range extends north and west from the Colorado River. It extends from the desert areas of California south of the San Joaquin Valley, eastward across the Mojave Desert into southern Nevada, the extreme southwestern corner of Utah (i.e., the Beaver Dam Slope), and the extreme northwestern corner of Arizona, as well as southeast across the Colorado Desert to the Colorado River. The Desert Renewable Energy Conservation Plan (DRECP) Area supports

individuals attributed to Agassiz's desert tortoise, or the Mojave population, as shown in Figure SP-R4.

Figure 1. Distribution of Agassiz's desert tortoise and Morafka's desert tortoise (Murphy et al. 2011, contained in USGS 2011).

Distribution and Occurrences within the Plan Area

Historical

The historical distribution of the desert tortoise (including both the currently recognized Agassiz's and Morafka's desert tortoise species) appears to be mostly the same as today. However, some authors indicate its range may once have been broader at the end of the Pleistocene, extending as far east as Texas and to coastal Southern California in the west. It is hypothesized that its range contracted to its current size about 8,000 years ago (Moodie and Van Devender 1979; Van Devender and Moodie 1977). Native Americans used the tortoise for a variety of purposes, including food, ceremonial uses, medicinal uses, household (utensil) uses; it also figured prominently in Native American mythology and symbolism (Schneider and Everson 1989). There are 33 historical (i.e., before 1990) occurrence records in the Plan Area (Dudek 2013) (Figure SP-R4).

Recent

Although in areas of extreme dryness its numbers are much reduced, the Agassiz's desert tortoise (hereafter tortoise or desert tortoise) is found throughout the DRECP Plan Area. For instance, the tortoise is mostly absent from the valley floor of the very hot, dry Coachella Valley, including the valley west of the Plan Area, but instead can be found on the lower slopes of the surrounding desert mountains (Coachella Valley Conservation Commission 2007). Additionally, some studies indicate that the tortoise may utilize available local habitat in a non-random fashion, perhaps focusing its activities in high plant diversity and low sand abundance areas (Baxter 1988; Duda et al. 2002; Wilson and Stager 1992). There are 1,642 recent (i.e., since 1990) occurrence records the Plan Area (Figure SP-R4) (Dudek 2013).

Natural History

Habitat Requirements

The desert tortoise can be found in a wide variety of habitats, such as alluvial fans, washes, canyons, and saltbush plains (Coachella Valley Conservation Commission 2007; Woodbury and Hardy 1948; Lovich and Daniels 2000; USFWS 1994) (Table 1). Whereas most tortoises in the Mojave Desert are usually associated with creosote bush (*Larrea tridentata*) scrub on alluvial fans and bajadas (USFWS 2011a), they can also be found in saltbush scrub (*Atriplex* spp.) (Stewart 1991) and even in some man-made structures, such as artillery mounds (Baxter 1988). Individuals in the Sonoran Desert are associated more with the low rocky slopes of the desert mountains (Schamberger and Turner 1986, Barrett 1990).

The presence of shrubs in tortoise habitat is extremely important. Shrubs not only supply shade for the tortoises during hot weather (Marlow 1979), but also their roots provide support and protection for tortoise burrows. For instance, near Twentynine Palms, California, 71% of desert tortoise burrows were associated with creosote bush, and desert tortoises avoided the only community without creosote bush (Baxter 1988). However, other investigators found that burrows were not significantly closer to creosote bush than random sites in areas with vegetation representing both Mojave

and Sonoran affinities. Burrows were significantly farther from yucca (*Yucca* spp.) than random sites (Lovich and Daniels 2000). In still another case, burrows were associated with Mojave yucca (*Yucca schidigera*) and catclaw acacia (*Acacia greggii*) even though these species were not particularly abundant (Burge 1978). Wilson et al. (1999) found that most juvenile burrows were associated with shrubs. These studies point out that utilization of shrubs varies with the location of the study site; nevertheless, shrubs provide important resources for the desert tortoise.

Several studies have also shown that edaphic (soil) conditions are important for desert tortoises. Tortoises spend up to 98% of their lives underground (Nagy and Medica 1986). Where soils are so sandy that they cannot support the roof of a burrow, tortoises are unlikely to utilize the area (Baxter 1988). In a multivariate analysis of tortoise abundance criteria, Weinstein et al. (1986) indicated that "soil digability" is a significant regression variable (i.e., this variable accounted for a significant amount of the variance in habitat use). Conversely, if a caliche horizon (a hardened deposit of calcium carbonate) is present, it may be so hard that tortoises cannot successfully burrow under it. For instance, at the Twentynine Palms Marine base, Baxter (1988) found that every "tank pit" supported tortoise burrows, most often located just under the hardpan.

Table 1. Habitat Characteristics of the Desert Tortoise within the Southwest (adapted and abridged from Germano et al. 1994)

Habitat Features	Western Mojave Desert	Eastern Mojave Desert	Sonoran Desert (Morafka's desert tortoise)
Occupied Habitat	Valleys, bajadas, hills	Valleys, bajadas, hills	Bajadas, rocky slopes
Substrate	Sandy loams to rocky	Sandy loams to rocky	Rocky
Vegetation	Low-growing sclerophyll shrubs	Low-growing sclerophyll shrubs	Low-growing to arborescent sclerophyll shrubs
Annual Plants	Mostly winter germinating	Mostly fall germinating, some summer germinating	Mostly summer-germinating

Foraging Requirements

Tortoises are herbivores; wildflowers, grasses, and in some cases, cacti make up the bulk of their diet (USFWS 2010; Woodbury and Hardy 1948). Some of the more common herbaceous species utilized by the tortoise include desert dandelion (*Malacothrix glabrata*), primrose (*Oenothera* spp.), gilia (*Gilia* spp.), desert marigold (*Baileya multiradiata*), and filaree (*Erodium* spp.). Additionally, tortoises may eat some grasses, such as Indian rice grass (*Oryzopsis hymenoides*) or galleta grass (*Hilaria rigida*), although the nutritional value may be less. Also, tortoises are known to eat some cacti such as prickly pear (*Opuntia mohavensis*), beavertail (*Opuntia basilaris*), and various cholla cacti (*Opuntia* spp.). Spring desert annuals and grasses are particularly important in that they supply tortoises with much needed water (USFWS 2010), which can be stored by the tortoises for long periods of time (Marlow 1979; Woodbury and Hardy 1948). In Twentynine Palms, California, desert tortoises were found in plant communities with high plant species diversity, such as washes and ecotones between communities (Baxter 1988). Although tortoises were captured more frequently in the diverse wash community—significantly more than expected based on a random distribution—this could be a result of higher visibility to the surveyors in these areas. Nevertheless, their burrows were also significantly closer to ecotones than a set of random points. The use of these high plant diversity areas may therefore be related to increased food availability or possibly the nature of the annual herbs found in these areas.

Reproduction

The desert tortoise breeds in the late summer and fall, before going into hibernation for the winter. Males will "joust" to establish loosely defined home ranges, but these can overlap and are not exclusive. Home range size can vary dramatically, from 10 to over 450 acres (USFWS 1994). Females begin breeding at about 15 to 20 years of age, and will store the male's sperm (Gist and Fisher 1993; Turner and Berry 1984). Egg laying occurs in the spring, but occasionally may also take place in the fall. Incubation is typically about 100 days, with the eggs hatching in the late summer and early fall. There is little or no parental care of the nest or the young. The sex of the offspring is determined by the incubation temperature; females being hatched at

higher ground temperatures (above 89°F) while males are hatched below this temperature (Spotila et al. 1994). Average clutch size is 4.5 eggs (Turner et al. 1984, 1986).

Spatial Behavior

Tortoise activity is focused on its home range, and is primarily determined by temperature (USFWS 1994). Nevertheless, some relocated tortoises have moved significant distances from their release point, including crossing major highways (Stewart 1991). Duda et al. (1999) found that tortoise home ranges tend to shrink during periods of drought compared to years of high rains. Following winter hibernation, tortoises become active as low temperatures abate in the spring months. During the spring, tortoises are active throughout the day, foraging on the fresh shoots of annual plants. But as the heat continues to increase into the summer months, tortoises are active only in the cooler morning, late afternoon, and evening hours. During the hot daytime temperatures, tortoises retreat to burrows to wait it out or, in some cases, will aestivate through the summer.

Ecological Relationships

The desert tortoise is a primary consumer; that is, they feed on plants. As such, they compete for vegetation resources with other primary consumers, such as the desert iguana (*Dipsosaurus dorsalis*), Gambel's quail (*Callipepla gambelii*), mourning dove (*Zenaida macroura*), pronghorn antelope (*Antilocapra americana*), and domestic cattle (*Bos taurus*). Adult tortoises are preyed on by few other animals; however, some may be taken by coyote *(Canis latrans*) and kit fox (*Vulpes macrotis*). Young tortoises are routinely preyed upon by kit fox and common raven (*Corvus corax*).

Desert tortoise burrows supply important shade and thermoregulatory resources for a variety of species, including many species of snakes, insects and spiders, and small mammals.

Population Status and Trends

Global: Declining (USFWS 2011a; Corn 1994; Bury and Corn 1995; Berry and Medica 1995; Woodman 2004)

S**tate:** Same as above
Within Plan Area: Same as above

According to the Revised Recovery Plan for the Mojave Population of the Desert Tortoise (*Gopherus agassizii*), the Mojave population occurs north and west of the Colorado River in the Mojave Desert of California, Nevada, Arizona, southwestern Utah, and the Colorado Desert in California (USFWS 2011a). Historic information for the Mojave population densities or abundance does not exist to provide a baseline for population trends (USFWS 2011a). Long-term study plots and other studies, however, suggest "appreciable declines" at the local level in many areas, and that the identified downward trend of the species in the western portion of the range at the time of the federal listing as threatened in 1990 was valid and is ongoing (USFWS 2011a). Results of studies in other parts of the Mojave population's range also are inconclusive, but suggest that declines are broadly distributed across the tortoise's Mojave Desert range (USFWS 2011a). In addition, specific management actions over a 23-year monitoring program have not demonstrated a positive effect on populations, although the life history of the species (i.e., delayed reproductive maturity, low reproductive rates, and relatively high mortality early in life) is such that rapid increases in populations are unlikely to be observed (USFWS 2011a).

Threats and Environmental Stressors

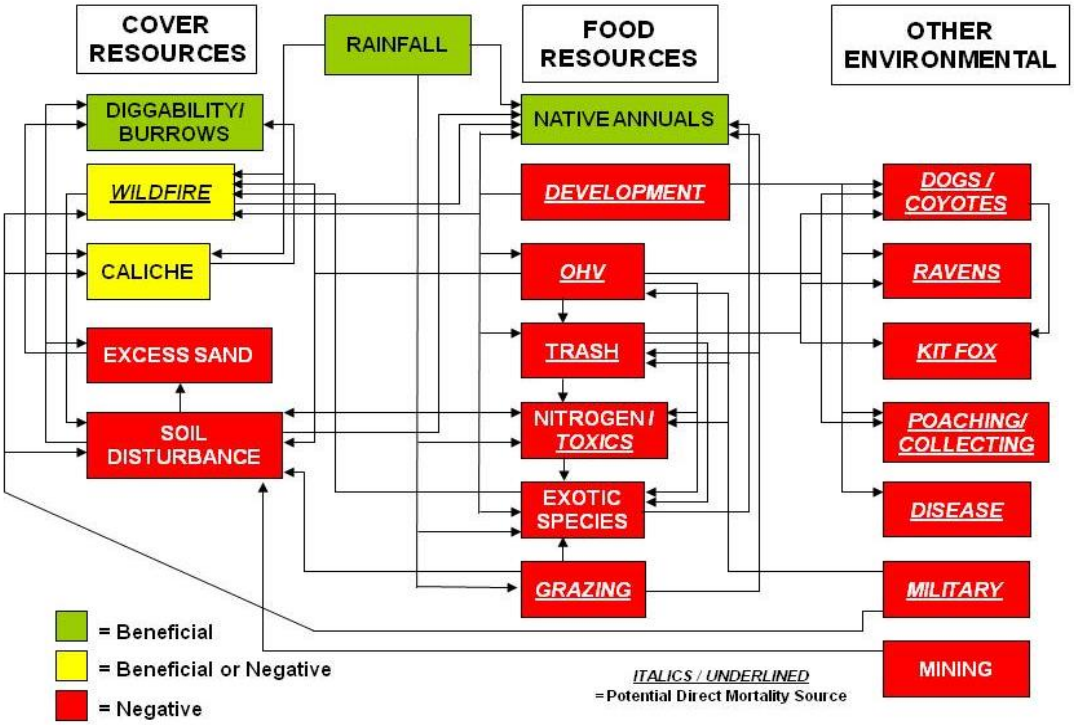
The desert tortoise is faced with a multitude of threats and environmental stressors to its survival. Many of these threats are synergistic (Tracy et al. 2004). Figure 2 presents a generalized conceptual model of some of the more important threats and stressors to the desert tortoise. For a detailed review of these threats and stressors, please see USFWS (2011a) and Boarman (2002). Chief among these threats are:

- Predation;
- Habitat loss and fragmentation;
- Disease;
- Other human activities (e.g., agriculture, fire, landfills, grazing, military activities);

- Off-highway vehicle (OHV) use;
- Collecting; and
- Invasive species.

Figure 2. Example of a generalized conceptual model of tortoise threats and stressors.

Predation: The desert tortoise is subject to predation from several



feral dog (*Canis familiaris*) (Evans 2001), coyote, and kit fox (Bjurlin and Bissonette 2001), although the precise magnitude of impacts remain unclear (Turner et al. 1987). However, the majority of predation occurs on incubating eggs and young tortoises whose shells are still soft. In addition, predation of the young by the common raven is becoming increasingly important (Campbell 1985; Berry 1985; Boarman 1993; Kristan and Boarman 2003). Although a "natural predator," raven populations in the Mojave Desert increased by 1,000% between 1968 and 1992. This increase is sometimes attributed to the increase in landfills (Engel and Young 1992), but it could also be related to the increase in roads, providing roadkill for this highly opportunistic species (Boarman 1993; Boarman and Berry

1995). Increased predation by coyotes has been shown to be a major factor affecting the success of a large-scale relocation of desert tortoises at Fort Irwin (Berry et al. 2011).

Habitat Loss and Fragmentation: Habitat loss and fragmentation are often considered one of the most important factors in reducing tortoise numbers (U.S. Bureau of Reclamation 2008; USFWS 1994; Berry and Burge 1984). Residential and infrastructure development, as well as infrastructure improvements, have the effect of directly reducing available tortoise habitat, but also introduce a number of indirect effects, such as attractants to ravens and coyotes and invasive plant species. Further development and associated roads act as barriers to tortoise movement (as well as sources of direct mortality) that fragment populations into smaller subpopulations. Generally speaking, models have shown that populations of species that are physically isolated are more likely to be extirpated by stochastic, demographic, and/or genetic consequences (Gilpin and Soulé 1986.)

Disease: Major threats to the continued existence of the desert tortoise come from several diseases (Jacobson 1994). Principal among these are upper respiratory tract disease caused by the bacteria, *Mycoplasma agassizii* and *M. testudineum* (Berry 1997; Brown et al. 1999; USFWS 2011a), and cutaneous dyskeratosis, a shell disease (Jacobson et al. 1994; Homer et al. 1998). It is often thought that these diseases were introduced into native populations by the release of infected pets back into the wild (Boarman 2003; Coachella Valley Conservation Commission 2007; USFWS 2011a; Johnson et al. 2006). From 1979 to 1992, the population of tortoises at the Desert Tortoise Natural Area, near Mojave, California, decreased by 76% (Berry 1997; Hardenbrook and Tomlinson 1991), with the last 5 years attributed to disease.

OHV Use: For decades, the use of OHVs in the desert has continued to increase in frequency. This use includes a wide spectrum of activities, ranging from occasional personal use for access, to other activities (e.g., camping, rock hounding, photography, research), to large organized competitive events. In addition to direct mortality by crushing, the list of potential impacts from OHV use is great; it includes destruction and degradation of vegetation (forage), soil compaction, and the destruction of cryptogamic soils, but also facilitation of erosion (Adams et al. 1982; Berry 1990; Berry et al.

1994; Bury and Luckenbach 1986; Davidson and Fox 1974; Vollmer et al. 1976). With the increase in backcountry visitation, other indirect impacts can increase, such as the introduction of invasive plants, increased trash dumping (which can attract common ravens, coyotes, and feral and pet dogs), increased fires, and the introduction of pets (USFWS 1994).

Collecting: Desert tortoises are often collected as pets. Stubbs (1991) discusses the general aspects and causes of human collecting of wildlife. Data for this phenomenon are mostly anecdotal; however, Stewart (1991) documented the removal and possible killing of tortoises that were radio-collared (see also Berry 1990). As mentioned previously, re-release of captured tortoise back into the wild is often cited the source of introduction of disease into native populations (USFWS 1994). This release of pet tortoises can also result in the increase in competition for scarce resources with resident native tortoises, as well as possibly serving as a source of genetic contamination. It remains unclear as to the magnitude of this threat (Boarman 2002).

Invasive Species: The Plan Area has been subject to invasion by numerous invasive plant species (Brooks 1998; Boarman 2002). Principal among these are non-native annual grasses (e.g., *Bromus* spp., *Schismus* spp.), tamarisk, and, more recently, invasive Sahara mustard (*Brassica tournefortii*). Although these introduced species may serve as some forage for tortoises, their nutritional value is likely less than native species. These species colonize rapidly following fires or other ground disturbances (Brown and Minnich 1986; Davidson and Fox 1974; Hobbs 1989), competing against native annuals and perennial seedlings for the sparse resources, as well as in some cases, preventing movement of some species. In some areas, native vegetation has been replaced by essentially monospecific stands of these invaders (see Brooks 1998, 2000).

Other Human Activities: Numerous other human activities affect desert tortoise, many of which are interrelated. Agriculture affects desert tortoises through conversion of habitat into mostly unsuitable uses (Boarman 2002, 2003) and can introduce invasive species and toxins into the environment. Fire can impact tortoises through direct mortality (Homer et al. 1998) but also by the type-conversion of native

habitat to non-native grasslands and weedy forbs. These grasses and forbs can, in turn, increase flashy fuel loads and fire frequency, exacerbating and increasing the frequency of the problem (Esque et al. 1994; Jacobson 1994). Landfills have the direct effect of usurping sometimes large areas of available habitat, but their primary impact to tortoises results from an increase in the number of predators (coyotes, common ravens, feral dogs) they can attract (Boarman 1993, 2003; Engle and Young 1992). Grazing can reduce forage available to desert tortoises (Nicholson and Humphreys 1981; USFWS 1994), as well as occasionally killing them outright or destroying nests by trampling (Jacobson 1994). Grazing can also increase the presence of non-native invasive species (Brooks 1998). However, quantitative data on the actual direct impacts of grazing, both cattle and sheep, are generally lacking (Boarman 2002). Military activities can result in direct mortality of tortoises by crushing (Baxter and Stewart 1990; Stewart and Baxter 1987), as well as the loss and degradation of habitat and the collapse of burrows and nests (USFWS 1994).

Conservation and Management Activities

Following the listing of the desert tortoise, the Desert Tortoise Management Oversight Group (Oversight Group) was established in 1988. The initial purpose of the Oversight Group was to coordinate agency management and planning, and to begin implementation of management strategies on (primarily) Bureau of Land Management (BLM) land (USFWS 2011a). In addition to BLM staff, USFWS staff was initially included, but the Oversight Group was later expanded to include representatives from the Department of Defense, U.S. Geological Survey, and the National Park Service. The purpose of the Oversight Group was to serve as a clearinghouse of the various agencies' tortoise management plans and implementation, identify data gaps and threats, and provide review of ongoing research into the desert tortoise (USFWS 2011a).

In 2003, USFWS, following recommendations of a General Accounting Office (GAO) report (GAO 2002), created the Desert Tortoise Recovery Plan Assessment Committee, which was empowered to review the successes and failures of the initial 1994 recovery plan. This report was completed in 2004 (Tracy et al. 2004). Generally the report found

that the recovery plan of 1994 was serving its function, but that the plan needed to be revised based upon new knowledge of desert tortoise biology, ecology, genetics, the previously unappreciated synergistic nature of the multiple threats, and advances in scientific techniques, which had been elucidated over the previous decade. The report also echoed the conclusion of the GAO report that called for a concerted, coordinated effort by the various agencies, especially in the identification and interpretation of basic desert tortoise research. To this end, USFWS established the Desert Tortoise Recovery Office (DTRO) in 2004. Since that time, the DTRO has served as the principal clearinghouse for research and monitoring of the desert tortoise north and west of the Colorado River (USFWS 2011a). It also coordinates activities of the Oversight Group, and [later] the Desert Manager's Group, as well as other agencies and scientists working on the tortoise (USFWS 2011a). The DTRO also established a desert tortoise science advisory committee in 2005 to provide scientific advice on recovery tasks, ensuring a sound scientific basis for their results and conclusions.

In 1995, the Desert Manager's Group was established as the forum for government agencies to address and discuss issues of common concern. Not just focused on the desert tortoise, the Desert Manager's Group seeks to provide a forum for cooperative management that provides "… greater operational efficiency, enhances resource protection, and the public is better served" (Desert Manager's Group 2005), but nonetheless has produced a 5-year plan related to several tortoise issues (USFWS 2011a).

Based on recommendations in the recovery plan assessment (Tracy et al. 2004), the goals of management for the desert tortoise are:

- Maintain self-sustaining populations of desert tortoises within each recovery unit into the future;
- Maintain well-distributed populations of desert tortoises throughout each recovery unit; and
- Ensure that habitat within each recovery unit is protected and managed to support long-term viability of desert tortoise populations.

The revised recovery plan (USFWS 2011a) calls for a revision of the existing recovery plan (USFWS 1994) with the following goals:

1. Develop, support, and build partnerships to facilitate recovery: The revised recovery plan proposes to establish recovery implementation teams to coordinate and evaluate management and monitoring at a recovery unit level. The recovery implementation teams will also be charged with providing education and outreach activities. Protect existing populations and habitat, instituting habitat restoration where necessary: The revised recovery program calls for increased protection of desert tortoises within "tortoise conservation areas" defined as, "… desert tortoise habitat within critical habitat, desert wildlife management areas, areas of critical environmental concern, Grand Canyon–Parashant National Monument, Desert National Wildlife Range, National Park Service lands, Red Cliffs Desert Reserve, and other conservation areas or easements managed for desert tortoises," or areas further identified by the individual recovery implementation teams. The plan also indicates the importance of recognizing that areas outside the conservation areas may affect what happens within them and recommends a broader outlook toward implementation through interagency cooperation and coordination.
2. Augment depleted populations in a strategic manner: The revised recovery plan calls for the augmentation of depleted or extirpated populations of the desert tortoise. This augmentation should be completed as an adaptive management strategy, focusing its implementation on answering not only important questions regarding the success of relocation techniques, but also those of understanding threats and stressors.
3. Monitor progress toward recovery: A new approach toward monitoring is proposed that not only assesses the status of desert tortoise populations (at 5-year intervals), but also includes multidimensional monitoring of such variables as threats, habitat quality, and changes that could be related to climate change. Monitoring will focus on those metrics directly related to recovery criteria.

4. Conduct applied research and modeling in support of recovery efforts within a strategic framework: Similar to No. 4 (above), the revised plan indicates a need to fill data gaps in tortoise biology and ecology through applied adaptive research activities. In particular, the plan identifies the need to investigate the synergistic nature of human threats to the tortoise, how they interrelate, and how these in turn affect tortoise abundance.
5. Implement a formal adaptive management program: Based on conceptual models (see Figure 2 as an example), and using data gathered from the implementation of the above programs, the revised recovery plan calls for the formal structuring of an adaptive management program, coordinated through the DTRO, to integrate the results of the various adaptive management experiments.

The revised recovery plan also calls for a revision of the desert tortoise recovery units. Based on recent genetic work (Murphy et al. 2007; Hagerty and Tracy 2007), it is proposed to redefine the units from an initial six to five units. The principal changes are results of combining and expanding the previous northern Colorado and eastern Colorado units into one (i.e., Colorado Recovery Unit, Figure 4), a contraction of the Eastern Mojave Recovery Unit, an appurtenant expansion of the Northeastern Recovery Unit, and a contraction of the southern extreme of the Western Mohave Recovery Unit in the vicinity of the Coachella Valley. Figure 3 shows the revised recovery units.

Figure 3. Revised recovery units from draft revised recovery plan (USFWS 2011a).

Data Characterization

The desert tortoise has supported a long history of research. Since 1976, many of these data and results have been presented annually at the yearly symposium of the Desert Tortoise Council (Beaumont, California). Papers have addressed virtually every aspect of desert tortoise ecology, physiology, and behavior. In spite of the plethora of reports, USFWS (2008) states, "However, despite clear demonstration that these threats impact individual tortoises, there are few data available to evaluate or quantify the effects of threats on desert tortoise populations. While current research results can lead to predictions about how local tortoise abundance should be affected by the presence of threats, quantitative estimates of the magnitude of these threats, or of their relative importance, have not yet been developed. Thus, a particular threat or subset of threats with

discernible solutions that could be targeted to the exclusion of other threats has not been identified for the desert tortoise."

Management and Monitoring Considerations

Although specific management of the desert tortoise in the Plan Area will likely be site-specific (e.g., fencing locations, patrol routes, artificial burrow locations), particularly as each site relates to anthropocentric activities either on the site or nearby, generally, overall management should include the following activities, all of which should be coordinated with the USFWS Desert Tortoise Recovery Office and the respective recovery implementation team:

- Establishment of a series of occupied preserves of native tortoise (and other species) habitat using the best currently understood principles of conservation biology, such as, but not limited to, connectivity and movement corridors, distinct genetic varieties, and reserve size.
- Creation of educational programs to inform the public about the tortoise, other desert species, and desert ecosystems; in particular, supply of information regarding the dangers of releasing pet tortoises back into the wild and the effects of trash dumping and OHV activities.
- Creation of enforcement programs to ensure the integrity of the preserve system to minimize levels of threats and stressors.
- Funding of continued research into the precise nature and effects of threats and stressors of the desert tortoise. This offers the best avenue for long-term management by furthering understanding of the ecological relationships of the tortoise, thereby making management decisions more focused and efficacious.
- Establishment of ongoing adaptive management programs to elucidate the effects of threats and stressors of the desert tortoise.
- Establishment of a repository for captured or sick tortoises to help prevent their release into the wild.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Agassiz's desert tortoise, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 12,642,923 acres of modeled suitable habitat for desert tortoise in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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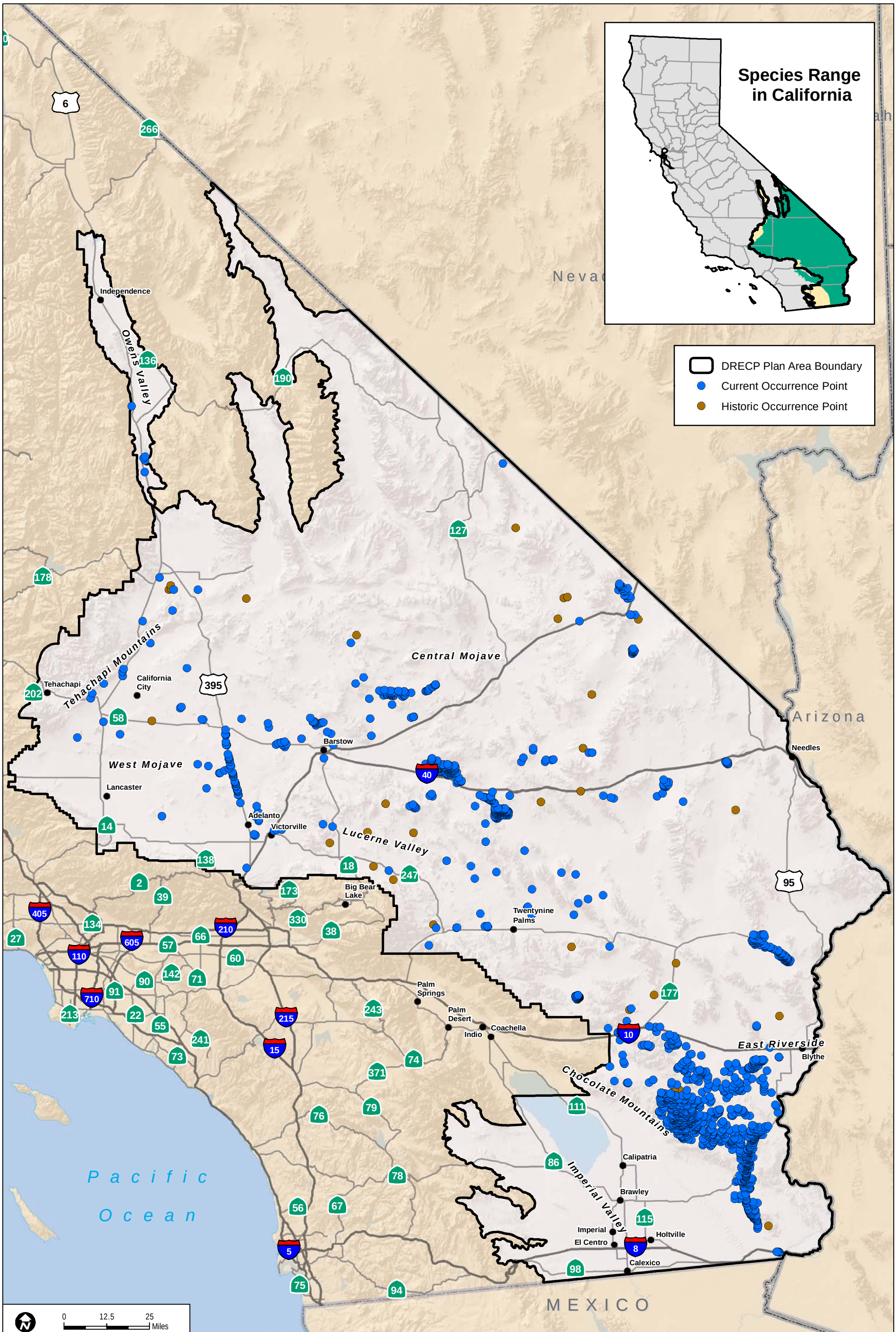

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-R01

Desert Tortoise Occurrences in the Plan Area

Flat-Tailed Horned Lizard (*Phrynosoma mcallii*)

Photo courtesy of Brock Ortega, Dudek.

Legal Status

State: Species of Special Concern
Federal: Bureau of Land Management Sensitive, U.S. Forest Service Sensitive
Critical Habitat: N/A
Recovery Planning: N/A (Note: A Flat-tailed Horned Lizard Rangewide Management Strategy [Flat-tailed Horned Lizard Interagency Coordinating Committee 2003] has been developed.)
Notes: The species has been proposed for listing by the U.S. Fish and Wildlife Service (USFWS) on four separate occasions (1993, 2001, 2005, 2010). On March 15, 2011, the USFWS published a proposed rule determining that the flat-tailed horned lizard does not require protection under the federal Endangered Species Act (ESA) (76 FR 14210–14286).

Taxonomy

The flat-tailed horned lizard (*Phrynosoma mcallii*) was first described in 1852 by Hallowell, and is one of eight recognized horned lizard species in North America (Flat-tailed Horned Lizard Interagency Coordinating Committee [ICC] 2003). The flat-tailed horned lizard is closely related to the Goode's horned lizard (*P. goodei*) and desert horned lizard (*P. platyrhinos*), which it may hybridize with where their ranges overlap (Jones and Lovich 2009). Both of these two species can be differentiated from the flat-tailed horned lizard by their shorter occipital horns and lack of a dark mid-dorsal stripe (Jones and Lovich 2009). Descriptions of the species' physical characteristics can be found in Stebbins (1954) and Rorabaugh and Young (2009).

Distribution

General

The northern range limit of the flat-tailed horned lizard is in the Coachella Valley and extends southeast to the Imperial and Borrego valleys and into Baja California, Mexico. The western limit of the species' range is Anza-Borrego Desert State Park in eastern San Diego County, and to the east they are found in Glamis and Ogilby northwest of Yuma, Arizona, and then into the lower Colorado subdivision of the Sonoran Desert in Arizona (Jones and Lovich 2009). (Figure SP-R5)

Distribution and Occurrences within the Plan Area

Historical

The flat-tailed horned lizard has one of the most restricted ranges of all North American horned lizards (Stebbins 1985). The historic range of the flat-tailed horned lizard in California was approximately 1.8 to 2.2 million acres, primarily in Imperial County, but also in central Riverside and eastern San Diego Counties (Flat-tailed Horned Lizard ICC 2003). The historic western boundary was formed by Fish Creek, Vallecito, and the Santa Rosa Mountains. In addition another valley of habitat stretches to the west beyond Ocotillo and Coyote Wells where Interstate-8 meets Highway 92. The southern extent stretched into the Yuha Basin, ending at the Sierra Juarez and Coyote mountains. The eastern extent of the flat-tailed horned lizard range extended to the Algodones Dunes and is limited by the Chocolate and Cargo Muchacho Mountains (Hodges 1997). There are 216 historical (i.e., before 1990) occurrences of flat-tailed horned lizard in the Plan Area and an additional 269 occurrences of unknown observation date (Figure SP-R5) (CDFW 2013; Dudek 2013).

Recent

About 50% of the flat-tailed horned lizard historic range in California has been lost due to urban and agricultural development (Flat-tailed Horned Lizard ICC 2003). However, the rate of habitat loss and fragmentation are not even across this species' range, with closer to more than 90% habitat loss in Riverside County. From a niche model

using abiotic variables, Barrows et al. (2008) estimated that within the Coachella Valley there was originally 32,164 hectares (79,479 acres) of potential habitat for the fringe-toed lizard. From this they calculated a 91% to 95% loss of potential habitat when considering current conditions that would render that potential habitat unsuitable (Barrows et al. 2008).

The current known range for flat-tailed horned lizard begins near the confluence of the San Gorgonio and Whitewater rivers in Riverside County, and extends south and east through the Coachella Valley into Imperial County. Flat-tailed horned lizard are found on both sides of the Salton Sea, extending west into Borrego Valley with small extensions into the lower portions of the Coyote Creek Watershed, around Clark Dry Lake, north of the Fish Creek Mountains and southwest along San Felipe Creek. They are found on the Carrizo Wash east of Bow Willow, and may be found within the Carrizo Badlands. Their range extends east across East Mesa and the Algodones Dunes to Pilot Knob Mesa. Though their range extends into Arizona, the California population is separated by the Chocolate Mountains, Cargo Muchacho Mountains and the agricultural development near Yuma, Arizona (Turner et al. 1980, Wright 2003, NatureServe 2011). There are 1,794 recent (i.e., since 1999) occurrences of flat-tailed horned lizard in the southern portion of the Plan Area (Figure SP-R5) (CDFW 2013; Dudek 2013).

Natural History

Habitat Requirements

Flat-tailed horned lizards occupy the hottest and most barren areas of the Sonoran Desert. Suitable habitat is characterized as stabilized sand dunes that fall within the creosote-white bursage series of Sonoran Desert Scrub community (Turner and Brown 1982; Jones and Lovich 2009). They also occur in loose, active sand dunes, although often at the dune periphery or in more stable regions within the active dune habitat. Historically they have been found in extremely active dune hummock habitats in the western Coachella Valley where they have now been extirpated. They tend to occur at higher densities in eolian habitats that are more stable than those preferred by fringe-

toed lizards (*Uma* spp.), but there is substantial overlap in the habitat occupied by these lizards (Barrows, pers. comm. 2012).

Flat-tailed horned lizard is primarily associated with fine, moderately active eolian sands (Barrows and Allen 2010). Barrows et al. (2008) included six soil classifications in the model used to identify potential distributions of flat-tailed lizard: Myoma fine sand 5–15% slope (MaD), Myoma fine sand 0–5% slope (MaB), Coachella fine sand 0–2% slope (CpA), Coachella fine sandy loam 0–2% slope (CsA), Niland sand 2–5% slope (NaB) (Soil Conservation Service 1980, cited in Barrows et al. 2008), and a previously mapped region of ephemeral surface sand availability (Barrows and Allen 2007a, cited in Barrows et al. 2008).

Flat-tailed horned lizards occur at elevations from below sea level to about 250 meters (820 feet) above mean sea level (Arizona Game and Fish Department 2003). They are found where the substrate is composed of fine sands or silica. They are also found in areas that lack windblown sands such as the saltbush flats north of the Salton Sea, and the badlands in the Yuha Basin and Borrego Valley (Flat-tailed Horned Lizard ICC 2003). Flat-tailed horned lizards do not normally occur in habitats characterized as rocky mountainous areas, new alluvial areas with sloping terrain, major dune systems, marshes and tamarisk-arrow weed thickets, and agricultural and developed areas (Turner et al. 1980).

Table 1. Habitat Associations for Flat-tailed Horned Lizard

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Sand or pavement, creosote-white bursage	Dispersal, refugia, breeding	Dispersal, breeding (all life stages)	fine, moderately active eolian sands	Flat-tailed Horned Lizard ICC 2003

Foraging Requirements

Flat-tailed horned lizard feed almost exclusively on harvester ants (*Pogonomyrmex* spp.), but opportunistically eat small beetles, caterpillars, and termites (Flat-tailed Horned Lizard ICC 2003). The percentage of ants in their diet is greater than other horned lizard

species and in one study was found to be 97% of the prey items found in flat-tailed horned lizard stomachs (Flat-tailed Horned Lizard ICC 2003).

Reproduction

Mating usually occurs in May and June, but may start in April when adult flat-tailed horned lizards emerge from hibernation. Clutch size and number is dictated by the abundance of resources, and during a typical year females will lay one clutch of 4 to 6 eggs. With favorable conditions the females lay two clutches per season. The first clutch emerges in July and the second emerges around September. Reproduction may be at least doubled in wet years as opposed to dry years (Grant 2005). In dry conditions only the late season clutch will be produced (Young and Young 2000). Females travel outside of their home range to excavate a deep (80 to 100 centimeters [32 to 39 inches]) burrow where the eggs are deposited just below the level where the sand becomes visibly moist (Young and Young 2000). Hatchlings emerge from July through October. Flat-tailed horned lizards typically reach sexual maturity within their second year (Flat-tailed Horned Lizard ICC 2003) but may breed in their first year (Barrows and Allen 2009). Their typical life span is four years, but they have been documented to live up to six years (Flat-tailed Horned Lizard ICC 2003). This species has a relatively low mean longevity and extremely low reproductive rates relative to other Phrynosomatids. This combination renders this species extremely vulnerable to local extinctions over fairly quick time periods if habitats are fragmented or compromised with anthropogenic structures and activity (Barrows 2012, pers. comm.; Barrows and Allen 2009).

Table 2. Key Seasonal Periods for Flat-tailed Horned Lizard

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding				✓	✓	✓	✓	✓	✓	✓	✓	
Adult Hibernation	✓	✓								✓	✓	✓

Sources: Flat-tailed horned lizard ICC 2003; Barrows 2012, pers. comm.

Activity Patterns and Movement

Flat-tailed horned lizards are most active in the spring and fall, when they are active on the surface most hours of the day. During this period they are also active on the surface through the night (Flat-tailed Horned Lizard ICC 2003). During the increased summer temperatures their activity pattern shifts to two periods, morning and evening (Flat-tailed Horned Lizard ICC 2003). The optimum air temperature range for active flat-tailed horned lizards appears to be 35.2^{0}C to 40.2^{0}C (95.4^{0}F to 104.4^{0}F). They seek refuge in burrows or under the sand when daytime surface temperatures exceed 41.0^{0}C (105.8^{0}F) (Wright 2002; Wone and Beauchamp 2003).

Adult flat-tailed horned lizard are obligatory hibernators, spending most of the winter months (mid-October to mid-February) in burrows 5 to 10 centimeters (2 to 4 inches) below the surface (Flat-tailed Horned Lizard ICC 2003). Juvenile activity is also reduced during the winter, but they are occasionally seen foraging on warm winter days. It is thought that due to their smaller size they are not able to maintain a sufficient amount of fat reserves to remain in hibernation through the winter (Muth and Fisher 1992).

Home ranges for flat-tailed horned lizards can vary by population, sex, size of the individual, climatic conditions, or density of lizards, but typically are in the range of 1 to 10 acres, but can much larger at times. In some populations it is thought that flat-tailed horned lizard do not permanently maintain distinct home ranges, but rather shift their spatial use area over time (Flat-tailed Horned Lizard ICC 2003). Home ranges appear to vary in relation to resource conditions and sex. On study site near Yuma, Arizona Young and Young (2000) found that mean home range sizes for males was 6.2 acres during a dry year and significantly larger at 25.5 acres during a wet year. In contrast, mean female home ranges were 3.2 acres in a dry year and relatively the same at 4.7 acres in a wet year. This study also observed a wide variation in movement patterns among individuals, with a few home ranges estimated at greater than 85 acres.

Table 3. Movement Distances for Flat-tailed Horned Lizard

Type	Distance/Area	Location of Study	Citation
Mean Home Range	0.12 acre	Yuha Desert MA	Turner and Medica 1982
	6.7 acres	West Mesa MA	Muth and Fisher 1992
	8.8 acres	Yuma Desert MA	Miller 1999
Male mean annual home range	1.7-25.5 acres	Yuma Desert	Young and Young 2000
Female mean annual home range	2.4–12.6 acres	Yuma Desert	Young and Young 2000

Ecological Relationships

Of their known natural predators round-tailed ground squirrel (*Spermophilus tereticaudus*) and the loggerhead shrike (*Lanius ludovicianus*) were highlighted as major predators (76 FR 14210–14268). Other native predators include kestrels and roadrunners. These predators occur naturally though recent scientific literature suggests that the populations of some of these predators are now higher as a result of manmade changes to the landscape, resulting in increased predation of flat-tailed horned lizards localized near developed areas (76 FR 14210–14268). In addition, feral dogs and cats can prey on flat-tailed horned lizard. Recent studies have found a clear negative impact on flat-tailed horned lizard presence to at least 450 meters (1,476 feet) away from disturbance (Young and Young 2005).

Flat-tailed horned lizard has a relatively low mean longevity and extremely low reproductive rates relative to other Phrynosomatids. This combination renders this species extremely vulnerable to local extinctions over fairly quick time periods if habitats are fragmented or compromised with anthropogenic structures and activity.

Population Status and Trends

Global: Vulnerable (NatureServe 2011)
State: Imperiled (NatureServe 2011)
Within Plan Area: same as above

There are three regionally descriptive populations of flat-tailed horned lizard in California: Coachella Valley; the west side of the Salton Sea/Imperial Valley; and the east side of the Imperial Valley (NatureServe 2011; 76 FR 14214). The population in the Coachella Valley is divided into two segments by I-10. The two populations within the Imperial Valley are divided by I-8 and the Coachella Canal into four segments (Algodones Dunes, East Mesa, West Mesa/Anza Borrego, and Yuha) (Wright 2002). As discussed above, about 50% of the flat-tailed horned lizard historic range in California has been lost due to urban and agricultural development (Flat-tailed Horned Lizard ICC 2003). Most of this habitat conversion has occurred in the Imperial Valley between the Salton Sea and the U.S./Mexican border. However, the USFWS determined that current threats to the species identified in the 1993 proposed rule for listing the species as endangered are not as significant as formerly believed and available data do not indicate the species is likely to become endangered in the foreseeable future throughout all or a significant portion of its range (76 FR 14210-14286).

Threats and Environmental Stressors

The major identified threats to this species are habitat fragmentation and population isolation, agricultural development, urbanization, OHV use, highways, canals, railroads, military activities, utilities, predation, mining and mineral material extraction, geothermal power development, oil and gas development, wind turbines, landfills, exotic plants, fire, pesticide use, land disposal, cattle grazing, and other ground disturbance activities (Flat-tailed Horned Lizard ICC 2003; 76 FR 14223). Unregulated border patrol activities and related infrastructure development are also threats (Barrows and Allen 2009; Barrows 2012, pers. comm.). On March 15, 2011 the USFWS published the proposed rule for their determination that the flat-tailed horned lizard does not require protection under the federal ESA (76 FR 14210–14286). The proposed rule included an evaluation of potential

current threats, including agricultural and urban development, energy generation facilities, invasive plants, OHV use, military training, overutilization (e.g., collecting), and disease and predation. Generally, the USFWS concluded that while some level of threat to flat-tailed lizard and its habitat still exists from these factors, the level of threat is not substantial and does not justify listing of the species (76 FR 14210–14286). Nonetheless, these factors should still be considered threats to consider in the DRECP.

In a study examining boundary processes between natural and anthropogenic desert landscape the flat-tailed horned lizard demonstrated an unambiguous negative response to the anthropogenic habitat edges (Barrows et al. 2006). This effect was likely a result of road avoidance or road associated mortalities and predation from birds that may occur more often or be more abundant along habitat edges given the greater availability of resources in suburban areas (Barrows et al. 2006).

Conservation and Management Activities

On June 7, 1997, a Conservation Agreement, deemed a long-term agreement by its signatories, was signed by several federal and state agencies to implement the Flat-tailed Horned Lizard Rangewide Management Strategy (RMS) (updated in 2003). The following agencies are signatories to the Conservation Agreement:

- USFWS, Region 1
- USFWS, Region 2
- BLM, California State Office
- BLM, Arizona State Office
- Bureau of Reclamation, Lower Colorado Region
- U.S. Marine Corps Air Station, Yuma
- U.S. Naval Air Facility, El Centro
- Arizona Game and Fish Department
- California Department of Fish and Game
- California Department of Parks and Recreation.

The purpose of the RMS is to provide guidance for the conservation and management of the habitat for flat-tailed horned lizard (Flat-tailed Horned Lizard ICC 2003). The RMS identifies five Management Areas (MAs)–four in California and one in Arizona–that are to be maintained and managed in perpetuity. The four MAs in California are West Mesa, East Mesa, Yuha Desert, and Borrego Badlands (Anza-Borrego Desert State Park and Ocotillo Wells State Off-Highway Vehicle Area). The BLM, in coordination with the U.S. Navy manages the West Mesa and East Mesa MAs. BLM also manages the Yuha Desert MA. The California Department of Parks and Recreation manages the Borrego Badlands MA.

The Conservation Agreement remains in effect today, and the RMS continues to be implemented by all Conservation Agreement signatory agencies. As of 2009, the total management area is approximately 485,000 acres, of which 458,759 acres (95%) are under signatory ownership (76 FR 14217). Also, as of 2009, approximately 424 acres (0.09%) of the management area has been approved for development (76 FR 14217).

The RMS requires that an annual report be prepared by the Interagency Coordinating Committee to monitor plan compliance (Flat-tailed Horned Lizard ICC 2009).

The RMS calls for the following nine planning actions:

- Planning Action 1 – Delineate and designate five flat-tailed horned lizard MAs and one flat-tailed horned lizard Research Area.
- Planning Action 2 – Define and implement management actions necessary to minimize loss or degradation of habitat.
- Planning Action 3 – Within the MAs, rehabilitate damaged and degraded habitat, including closed routes and other small areas of past intense activity.
- Planning Action 4 – Attempt to acquire through exchange, donation, or purchase from willing sellers all private lands within MAs.
- Planning Action 5 – Maintain or establish effective habitat corridors between naturally adjacent populations.

- Planning Action 6 – Coordinate activities and funding among the signatory agencies with Mexican agencies.
- Planning Action 7 - Promote the Strategy through law enforcement and education.
- Planning Action 8 – Encourage and support research that will promote the conservation of flat-tailed horned lizards or desert ecosystems and will provide information needed to define and implement necessary management actions effectively.
- Planning Action 9 – Continue inventory and monitoring.

Every year the ICC reports on the progress of the nine planning actions. These reports, which are current to December 31, 2008, can be found on the Arizona USFWS website (http://www.fws.gov/southwest/es/arizona/Flat.htm).

The northern range of flat-tailed horned lizard, where habitat has been reduced to 3 to 4% of its original extent within the Coachella Valley, falls within the Coachella Valley Multiple Species Habitat Conservation Plan (CV MSHCP). The flat-tailed horned lizard is a covered species in the CV MSHCP, which would protect and manage approximately 44.5% of the remaining habitat. As of 2009, 94% of the projected protection of 4,219 acres habitat in the Thousand Palms conservation area and 34% of the projected protection of 5,134 acres in the Dos Palmas conservation area had been conserved (76 FR 14218).

Implementation of the Lower Colorado River Multi-Species Conservation Plan would have minor effect on the flat-tailed horned lizard because most the activities covered by the Plan are outside the range of the species and because the habitat is under the control of the Bureau of Reclamation, which is signatory to the Conservation Agreement discussed above (76 FR 14219). Impacts to approximately 128 acres of flat-tailed horned lizard habitat will be mitigated by acquisition of 230 acres in the Dos Palmas conservation area (76 FR 14219).

Data Characterization

Additional surveys are needed outside the RMS MAs to firmly delineate the boundaries on the exterior portions of flat-tailed horned lizard range in the United States (Foreman 1997).

Management and Monitoring Considerations

As mentioned above the Flat-tailed Horned Lizard RMS was developed in 2003 by local state and federal agencies to help manage for this species within its existing geographic range. The primary threat to this species is permanent habitat loss through urban and agricultural expansion (Young 2010). The threat of predation by both native and non-native predators is increased within several hundred meters along the edge between native intact habitat and agricultural development.

Currently management agencies are focused on monitoring population size as a means of detecting long term trends for flat-tailed horned lizards. It is the recommendation of Young (2010) that these monitoring efforts be altered to focus on covering larger areas utilizing scat surveys in place of current methods such as mark release recapture. Presence/absence surveys are much less expensive than obtaining population estimates, and will allow monitoring funds to be used in a manner that will reliably map and update the distribution of the species.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for flat-tailed horned lizard, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 624,072 acres of modeled suitable habitat for flat-tailed horned lizard in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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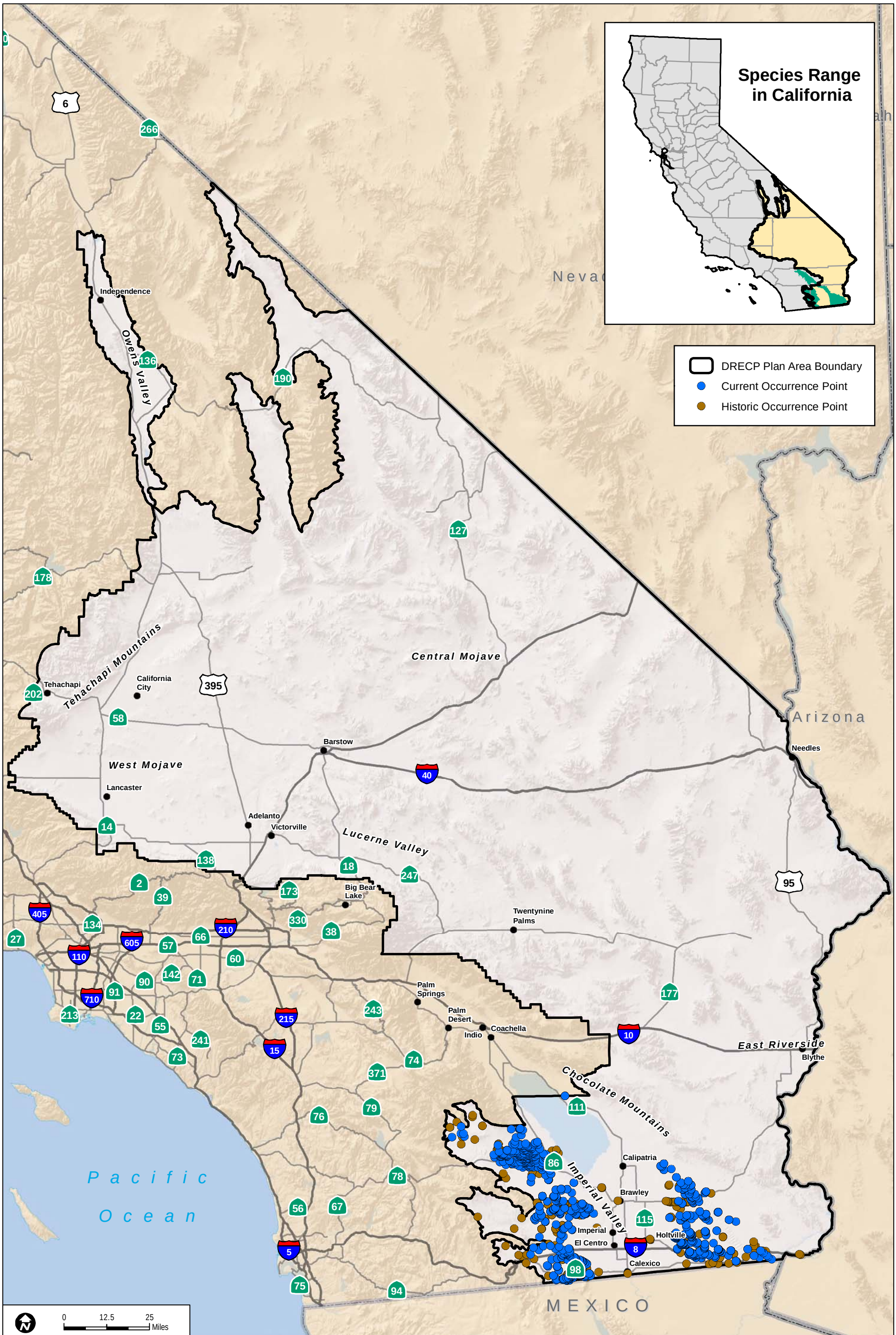

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-R02

Flat-tailed Horned Lizard Occurrences in the Plan Area

Mojave Fringe-Toed Lizard (*Uma scoparia*)

Legal Status

State: California Species of Concern
Federal: Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A
Notes: In 2006, a petition was filed to list the northern populations associated with the Amargosa River as a distinct population segment (DPS) under the Endangered Species Act. On October 4, 2011, the U.S. Fish and Wildlife Service (USFWS) published its 12-month finding, concluding that the Amargosa River population does not constitute a DPS and is not a listable entity (76 FR 61321–61330).

Taxonomy

The Mojave fringe-toed lizard (*Uma scoparia*) is a member of the *Phrynosomatidae* family of lizards that currently has 10 recognized genera occurring from southern Canada to western Panama (Reeder and Wiens 1996). The Integrated Taxonomic Information System (2011) currently recognizes six species of fringe-toed lizard in North America: the Mojave (*Uma scoparia*), the Yuma Desert (*U. rufopunctata*), the Chihuahuan Desert (*U. paraphygas*), the Colorado Desert (*U. notata*), the Coachella Valley (*U. inornata*), and the Coahuila Desert (*U. exsul*) fringe-toed lizard. The Amargosa River population has been identified as a potential DPS, although DNA sequencing found no evidence to support this (76 FR 61321–61330). Descriptions of the species' physical characteristics can be found in Stebbins (1954).

Distribution

General

The Mojave fringe-toed lizard is restricted to deposits of loose sand; as a result, its distribution is discontinuous throughout its range

(Fromer et al. 1983). The species is endemic to the Mojave and Sonoran deserts of Southern California and western Arizona. Within these regions, they are known to occur at more than 35 sand dune complexes in California and one in Arizona (Jarvis 2009). Figure SP-R6 depicts the range of this species in relation to the Desert Renewable Energy Conservation Plan (DRECP) Area.

Distribution and Occurrences within the Plan Area

Historical

Historically, this species was known to occur throughout the windblown sand areas in the following counties within the Plan Area: southern Inyo, San Bernardino, northern Los Angeles, and eastern Riverside. Within these counties, this species was known to occur within the present and historical river drainages and associated sand fields of the Mojave, Amargosa, and Colorado Rivers (Jarvis 2009). Outside of the Plan Area, they were known from La Paz County Arizona (Jones and Lovich 2009). Norris (1958) indicates that many of the major dune complexes are the result of reworking previous pluvial beach sands, and that fringing dunes adjacent to river systems may have been more continuous than the time of writing. Most date from the recent, while several others date from the Pleistocene. There are 18 historical (i.e., pre-1990) occurrences for Mojave fringe-toed lizard contained in the California Natural Diversity Database (CNDDB) and an additional 30 records with an unknown date of observation (CDFW 2013; Dudek 2013). These records are widely scattered throughout the Plan Area, generally in a region bounded on the west by the Palmdale area, on the northeast by the Black Mountains, on the east by the Turtle Mountains, and on the south by the Ford-Palen dunes area (Figure SP-R6).

Recent

There are 115 recent (i.e., since 1990) occurrences recorded in the Plan Area (Dudek 2013). Since 2006, Mojave fringe-toed lizards have been found in locations within the Amargosa River drainage that did not have any historic occurrence records. As described above, this species is currently found within more than 35 named and unnamed sand dune complexes within the three major river drainages in the

Plan Area: the Amargosa, Mojave, and Colorado rivers. Norris (1958) described 31 dune complexes. However, a more recent paper by Murphy et al. (2006) documents the extirpation of the species at four sites where they were previously reported (i.e., Harper and El Mirage dry lakes, Piute Butte, and Lovejoy Buttes). The named dune complexes are listed as follows with their associated river complex (76 FR 61321–61330).

Amargosa River

1. Ibex Dunes
2. Little Dumont Dunes
3. Dumont Dunes
4. Coyote Holes
5. Valjean Dunes

Mojave River

6. Hodge
7. Lenwood
8. Daggett
9. Yermo
10. Newberry Springs
11. Coyote Lake
12. Alvord Mountain
13. Cronese Lakes
14. Bitter Spring
15. Red Pass Dune
16. Silver Lake
17. Afton Canyon
18. Crucero
19. Rasor Road
20. Sands Siding

21. Devil's Playground – Kelso Dunes
22. Troy Dry Lake
23. Pisgah
24. Ludlow

Mojave and Colorado Rivers

25. Amboy Crater/Lava Field
26. Bristol Dry Lake
27. Cadiz Dry Lake
28. Dale Dry Lake East/West
29. Pinto Basin
30. Palen Dry Lake
31. Ford Dry Lake
32. Rice Valley.

Natural History

Habitat Requirements

The Mojave fringe-toed lizard is only found in and immediately around areas of the Mojave Desert that contain deposits of eolian, or fine windblown sands (Jones and Lovich 2009). These sands are typically associated with dunes, washes, hillsides, margins of dry lakes, and sandy hummocks between elevations of 90 and 910 meters (295 and 2,986 feet) (76 FR 61321–61330; Norris 1958; Stebbins 2003). Sand dune ecosystems, including their source sand and sand corridors, are necessary for the long-term survivorship of eolian sand specialists (Barrows 1996). Though sparsely vegetated, vegetation may include palo verde (*Parkinsonia florida*), mesquite (*Prosopis grandulosa*), creosote bush (*Larrea tridentata*), white bur sage (*Ambrosia dumosa*), indigo bush (*Dalea* sp.), sandpaper plant (*Petalonyx thurberi*), saltbush (*Atriplex* sp.), and numerous species of annuals (76 FR 61321–61330; Jarvis 2009).

Table 1. Habitat Associations for Mojave Fringe-Toed Lizard

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Windblown sands associated with creosote bush scrub generally associated with dune complexes, dry lake margins, and the base of hillsides	Dispersal, refugia, breeding	Dispersal, breeding	Windblown sands	Jones and Lovich 2009

Foraging Requirements

The Mojave fringe-toed lizard is best described as an opportunistic omnivore. They feed primarily on sand-dwelling insects, but will also feed on the flowers, leaves, and seeds of annual plants (Jarvis 2009). Juvenile Mojave fringe-toed lizards feed primarily on arthropods including ants, beetles, and scorpions. As they become adults, their diet shifts to include a more herbivorous diet (Jones and Lovich 2009). As is seen in many reptiles that live in arid environments, these lizards obtain most of their water from the insects and plants that they ingest (76 FR 61321–61330).

Reproduction

Sexual maturity is reached when individuals reach 65 to 70 millimeters (2.5 to 2.75 inches, snout-vent length, usually two summers after hatching [Jennings and Hayes 1994]). Mating typically occurs between April and late June (Table 2; 76 FR 61321–61330). Reproductive activity is highly dependent on the availability of sand-dwelling plants that grow in response to winter (October–March) rainfall (76 FR 61321–61330). Clutch size ranges from two to five eggs, but average two or three eggs (Miller and Stebbins 1964). During years with low rainfall females produce smaller clutch sizes, or none at all. Conversely, they may have multiple clutches in years with abundant rainfall (76 FR 61321–61330).

Table 2. Key Seasonal Periods for Mojave Fringe-Toed Lizard

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding				X	X	X						
Active			X	X	X	X	X	X	X	X		
Hibernation	X	X									X	X

Source: 76 FR 61321–61330.

Spatial Activity

Mojave fringe-toed lizards are most active from late spring through early fall, when they are active during the hotter periods of the day. According to Jones and Lovich (2009), their optimum body temperature is 37.3 degrees Celsius (99 degrees Fahrenheit), and they are rarely active when air temperatures are below 38 degrees Celsius (100 degrees Fahrenheit) or above 49 degrees Celsius (120 degrees Fahrenheit). They seek refuge in burrows or under the sand when daytime surface temperatures start to exceed 49 degrees Celsius (120 degrees Fahrenheit).

Home ranges for Mojave fringe-toed lizards vary greatly between sexes with adult males typically holding large (0.10 hectare or 0.3 acre) home ranges that are on average three times that of females. Both sexes display territorial behavior, although only males are known to defend their home ranges aggressively (Jones and Lovich 2009).

Dispersal of Mojave fringe-toed lizards is unlikely in the absence of nearby areas of windblown sands (76 FR 61321–61330). Within areas of active sand transport, sand dunes are highly dynamic and continually moving; in some cases, moving several meters per year. Movement between populations is poorly studied, although is likely limited by the natural movement of sands (Table 3). No specimen of Mojave fringe-toed lizard has been captured more than approximately 150 feet from windblown sand deposits (76 FR 61321–61330).

Table 3. Movement Distances for Mojave Fringe-Toed Lizard

Type	Distance/Area	Location of Study	Citation
Home Range Adult Male	0.10 hectare (0.3 acre)	Mojave	Kaufman n 1982
Home Range Subadult Male	0.02 hectare (0.05 acre)	Mojave	Kaufman n 1982
Home Range Female	0.034 hectare (0.084 acre)	Mojave	Kaufman n 1982

Ecological Relationships

Natural known predators of Mojave fringe-toed lizard include snakes, long-nosed leopard lizard (*Gambelia wislizenii*), greater roadrunner (*Geococcyx californianus*), burrowing owl (*Athene cunicularia*), loggerhead shrike (*Lanius ludovicianus*), hawks, American badger (*Taxidea taxus*), and coyote (*Canus latrans*) (Jones and Lovich 2009). Mojave fringe-toed lizard often uses burrows to escape predation. Burrowing rodents common in their habitat areas are round-tailed ground squirrel (*Spermophilus tereticaudus*), white-tailed antelope squirrel (*Ammospermophilus leucurus*), and various species of kangaroo rat (*Dipodomys* spp.) and pocket mouse (*Perognathus* spp.) (Fromer et al. 1983). In addition to predator avoidance, Mojave fringe-toed lizard use these rodent burrows for thermal protection during very high ambient temperatures.

Lizard species known to occur in habitats with similar characteristics as those preferred by the Mojave fringe-toed lizard include desert iguana (*Dipsosaurus dorsalis*), desert horned lizard (*Phrynosoma platyrhinos*), long-nosed leopard lizard, side-blotched lizard (*Uta stansburiana*), ornate tree lizard (*Urosaurus ornatus*), and zebra-tailed lizard (*Callisaurus draconoides*). Of these species, only zebra-tailed lizard appears to be a potential competitor of the Mojave fringed-toed lizard for food resources with Mojave fringe-toed lizard. These species are both insectivorous, approximately the same adult size, and likely select prey of similar size. Foraging behavior in the two species is similar, although not well documented (Fromer et al. 1983).

Population Status and Trends

Global: Vulnerable (NatureServe 2011)
State: Same as above
Within Plan Area: Same as above

The Mojave fringe-toed lizard is known to occur at more than 35 sand dune complexes in California and one in Arizona, all of which are naturally occurring within the species' historical range (76 FR 61321–61330; Norris 1958). Hollingsworth and Beaman (2001) state that although there is no published data suggesting a decline in population sizes of the Mojave fringe-toed lizard, enough urban development in the Mojave exists to cause concern that populations will be adversely affected. Bureau of Land Management (2002) states that there is no information about population trends. However, a more recent paper by Murphy et al. (2006) documents the extirpation of the species at four sites where they were previously reported (i.e., Harper and El Mirage dry lakes, Piute Butte, and Lovejoy Buttes).

Threats and Environmental Stressors

The loose windblown sand habitat that Mojave fringe-toed lizards rely on requires protection from direct and indirect disturbances (Barrows 1996). Direct disturbances to loose windblown sand habitat can include the use of off-road vehicles, the infestation and stabilization of dune sands by invasive exotic species (e.g., Sahara mustard [*Brassica tournefortii]*), and urban development. Direct disturbances to Mojave fringe-toed lizards include increases in local predators (e.g., common raven). Indirect disturbances to loose windblown sand habitat can include development of sand source areas, sand transport areas, and the use of sand barriers (e.g., sand fences) to control sand movement. It has been stated that this species is highly vulnerable to off-road vehicle activity and the establishment of windbreaks that affect how windblown sand is deposited (Stebbins 2003). The decline of the closely related Coachella Valley fringe-toed lizard is primarily attributed to habitat loss caused by urban development; disruption of the natural movement of sand caused by roads, windbreaks, and other man-made alterations; and off-highway vehicle use, which causes direct impacts to the species' habitat (Weaver 1981; Beatley 1994).

Conservation and Management Activities

Detailed research on the closely related Coachella Valley fringe-toed lizard conducted by Barrows (2006) suggested that the preservation of sand source corridors is critical for the long-term persistence of the species. The current management decisions being made in Coachella Valley should be used in informing management decisions and activities for the Mojave fringe-toed lizard.

Data Characterization

Although records from the California Natural Diversity Database (CDFW 2013) include 92 reports of the Mojave fringe-toed lizard within the Plan Area, there is surprisingly little information available on the current extent and population status of the species. The exception is the paper by Murphy et al. (2006) documenting the presence of the Mojave fringe-toed lizard at 21 sites (including one in Arizona) and the extirpation of the species at four sites. However, significant data are available for the Coachella Valley fringe-toed lizard (e.g., CVCC 2007). Regardless, there appears to be little data available about the effects of various stressors, including off-road vehicles, increased predator abundance, and invasive plant species, on the Mojave fringe-toed lizard.

Management and Monitoring Considerations

Management for the Mojave fringe-toed lizard includes not only the protection of occupied and potential habitat, but also the sources of transport avenues for the requisite sand. In discussing management for the closely related Coachella Valley fringe-toed lizard, the Coachella Valley Multiple Species Habitat Conservation Plan (CVCC 2007) indicates taking the following actions:

a. Control and manage impacts that degrade Coachella Valley fringe-toed lizard habitat, including fragmentation by roads, OHV use in protected habitat (except on designated routes of travel, if any), and other human disturbance.

b. Control human access to occupied habitat as necessary.

c. Evaluate the need as determined by monitoring for perimeter fencing to keep lizards inside conservation areas and away from roadways.

d. Identify actions to reduce impacts from, and control where feasible, invasive species if it is determined from monitoring results that there are impacts to Coachella Valley fringe-toed lizard habitat or populations.

e. Include measures to reduce the impacts to the lizards' food source, harvester ants, including aerial pesticide spraying (in coordination with the California Department of Department of Food and Agriculture) or introduction of exotic species (e.g., fire ants).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Mojave fringe-toed lizard, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 278,723 acres of modeled suitable habitat for Mojave fringe-toed lizard in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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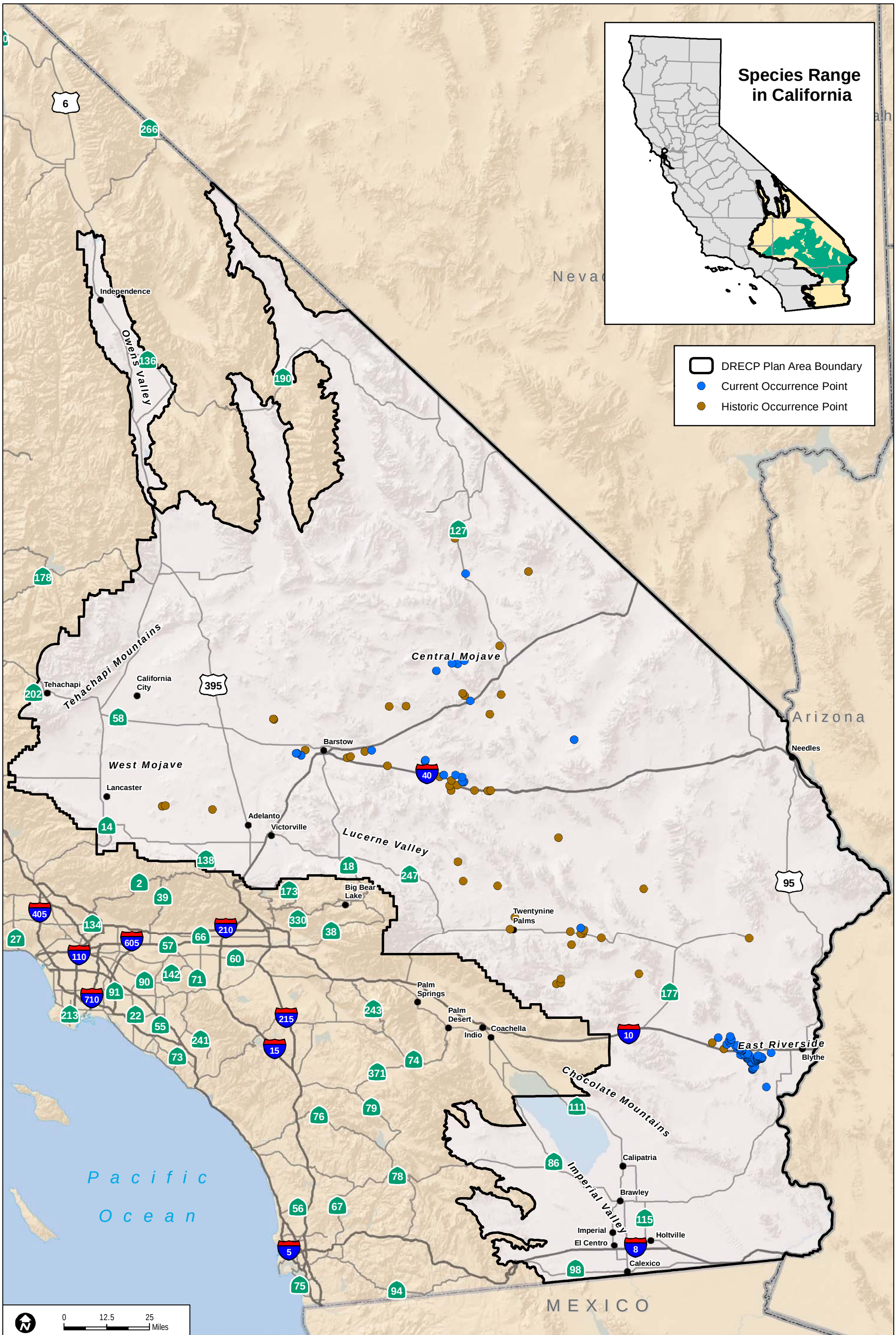

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-R03

Mojave Fringe-toed Lizard Occurrences in the Plan Area

Tehachapi Slender Salamander (*Batrachoseps stebbinsi*)

Courtesy of Gary Nafis.

Legal Status

State: Threatened
Federal: Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A
Notes: The recently completed (October 2011) U.S. Fish and Wildlife Service (USFWS) 12-month finding for Tehachapi slender salamander (*Batrachoseps stebbinsi*) to determine whether it should be federally listed as threatened concluded that a listing as threatened was not warranted (76 FR 62900–62926).

Taxonomy

The current description of the Tehachapi slender salamander (*Batrachoseps stebbinsi*) as a distinct species is relatively recent (Brame and Murray 1968). The taxonomy of Tehachapi slender salamander, however, is uncertain, and there is some evidence that Tehachapi slender salamander populations may represent two species. The existence of two species of *Batrachoseps* in the Tehachapi Mountains (in addition to the black-bellied salamander [*B. nigriventris*]) may have been recognized as early as 1858 (Wake and Jockusch 2000). Genetic work on speciation in *Batrachoseps* indicates a complex pattern of separation and contact among different species, which complicates the taxonomy of the genus. Wake and Jockusch (2002) examined the mitochondrial DNA gene cytochrome *b* for all 18 *Batrachoseps* species and several undescribed species and found that populations were more isolated in the past than they are now, indicating that there was some speciation occurring while separated. The recent contact and merging by male-mediated gene flow is confounding the genetic analysis. Hansen and Wake (2005) had suggested that the two populations centered in the Caliente Creek

area and in the Tehachapi Mountains, respectively, represent two distinct species based on differences in genetics, size, and coloration. However, in the recent 12-month finding of whether Tehachapi slender salamander should be federally listed as threatened, the USFWS evaluated the most recent available genetic and morphological information about differences between the two populations. USFWS's review included a personal communication with Hansen, who currently believes that there are insufficient differences between the two populations to classify them as separate species or subspecies (76 FR 62900–62926). Based on this review, USFWS concluded that the two populations of Tehachapi slender salamanders should be treated as a single species at this time. For the 12-month finding, USFWS assigned the Caliente Canyon and Tehachapi Mountains populations to two Distinct Population Segments (DPSs): the Tehachapi Mountains DPS and the Caliente Canyon DPS, which together constitute the entire range of the species (76 FR 62900–62926).

A description of the species' physical characteristics can be found on the CaliforniaHerps (2011) website or Stebbins (2003).

Distribution

General

The Tehachapi slender salamander is endemic to California and is reported to occur only in Kern County, although Morey (2005) indicates that the species could extend south into Los Angeles County. The California Natural Diversity Database (CNDDB) includes occurrences for elevations ranging from 1,610 feet in the Caliente Creek area to 5,575 feet in the Tehachapi Mountains (CDFW 2013) (Figure SP-R7).

The Tehachapi slender salamander occurs in two main DPSs that are geographically separated: (1) in the Caliente Creek drainage in the Paiute Mountains at the junction of the Sierra Nevada and Tehachapi mountains and (2) in the Tehachapi Mountains extending west to Fort Tejon State Park (76 FR 62900–62926).

The CNDDB contains a total of 20 records for Tehachapi slender salamander (CDFW 2013), all of which are documented from Kern

County. These occurrences were documented from 1957 to 2012 and all are considered extant, although their current presence has not been verified (CDFW 2013). Within the Caliente Canyon DPS, Tehachapi slender salamander has been recorded from 13 discrete localities at elevations of 1,610 to 6,000 feet (CDFW 2013).

HerpNet, a collaborative effort by natural history museums to establish a global network of herpetological collections data involving 64 institutions, includes 92 museum records for Tehachapi slender salamander. These records range from 1914 to 1979 (HerpNet 2010). Record localities include Live Oak Canyon in the Tehachapi Mountains; 6.3 miles southeast of Keene Store on U.S. 466; west of and southeast and southwest of Paris-Loraine/Loraine; along Caliente Creek Road; Fort Tejon; east of Caliente; northeast of Lebec at the mouth of Bear Trap Canyon; Caliente Canyon; near Caliente junction of Bealville Road and California Bodfish Road; and Tejon Canyon, 6.6 miles above Indian School (HerpNet 2010).

Distribution and Occurrences within the Plan Area

Historical

The Desert Renewable Energy Conservation Plan (DRECP) Area includes the eastern portion of the Tehachapi slender salamander's geographic range (Figure SP-R7). There is one historical (i.e., pre-1990) occurrence of the Tehachapi slender salamander in the Plan Area: a record from 1957 on private land from the Tehachapi Pass area near State Highway 58 (Dudek 2013). It was initially reported by Brame and Murray (1968) that the site was covered by a road, but as of 2008, the site was not covered by a road and remained in good condition, consisting of foothill pine (*Pinus sabiniana*), interior live oak (*Quercus wislizeni*), and California buckeye (*Aesculus californica*), as well as blue oak (*Quercus douglasii*) in open areas (CDFW 2013).

Recent

There are five recent (i.e., since 1990) records for the species in the Plan Area (Figure SP-R7): (1) a 2007 occurrence located in talus on the south side of Caliente Creek Road near the mouth of Big Last Chance Canyon (this site could also be considered historical because it was first

reported by Brame and Murray (1968)); (2) a 2009 occurrence located between Tollgate Canyon and Stevenson Creek about 7 miles north–northeast of State Highway 58; (3) a 2011 occurrence located in Silver Creek; (4) a 2011 occurrence located in Indian Creek; and (5) a 2011 occurrence in an unnamed canyon south of Indian Creek. The 2007 and 2009 occurrences are on Bureau of Land Management (BLM) lands (CDFW 2013), and the three most recent occurrences are on private land (76 FR 62900–62926; Dudek 2013). The three 2011 occurrences described in the USFWS 12-month finding extend the range of the Tehachapi slender salamander approximately 7 miles to the southeast of Caliente Canyon, but these are still considered part of the Caliente Canyon DPS (76 FR 62900–62926).

Natural History

Habitat Requirements

The Tehachapi slender salamander inhabits moist canyons and ravines in oak and mixed woodlands (see Table 1; CaliforniaHerps 2011). Vegetation in occupied habitat includes foothill pine, canyon live oak (*Quercus chrysolepis*), interior live oak, blue oak, Fremont cottonwood (*Populus fremontii*), western sycamore (*Platanus racemosa*), and California buckeye (Evelyn, pers. comm. 2012; Hansen and Wake 2005). At higher elevation sites, Tehachapi slender salamander has also been found with white fir (*Abies concolor*) (Evelyn, pers. comm. 2012). In more exposed areas of Caliente Creek, habitat includes California juniper (*Juniperus californica*), yucca (*Yucca* spp.), bush lupine (*Lupinus* spp.), and buckwheat (*Eriogonum* spp.). In the lower elevation Caliente Creek areas, the species is restricted to the lower margins of north-facing slopes and side canyons among granitic or limestone talus and scattered rocks (Hansen and Wake 2005). The species also occurs on north-facing slopes in the Tehachapi Mountains within talus piles and fallen wood (Hansen and Wake, pers. comm. 2008; Hansen and Wake 2005). The understory forb miner's lettuce (*Claytonia perfoliata*) is commonly found at occupied sites (Brame and Murray 1968).

During the moist periods of fall, winter, and spring precipitation, individuals seek cover under surface objects, especially rock talus (Brame and Murray 1968). Other substrates that may be used for cover include rocks, logs, bark, and other debris in moist areas

(CaliforniaHerps 2011), but they are primarily associated with talus (Hansen and Wake, pers. comm. 2008; Hansen and Wake 2005).

Specific habitat requirements for breeding or egg laying for this species are not well documented. Similar species lay their eggs underground or on moist substrates underneath or within surface objects, especially pieces of bark (Stebbins 1972).

It is unknown how or whether juvenile Tehachapi slender salamander habitat differs from that of adults. Juveniles are rarely found, which may indicate that hatching occurs in the spring, as surface activity declines, and that juveniles may remain underground (Hansen and Wake 2005).

Table 1. Habitat Associations for Tehachapi Slender Salamander

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Juniper woodland, Blue oak-foothill pine woodland, Mixed oak woodland, Riparian woodland	Primary habitat	Active and inactive season	North-facing talus and rocky slopes, 40% to 100% crown cover (species may be present even if the overall aspect of a slope faces east, south, or west, as long as there is a small draw that is shaded (usually north-facing)	Hansen and Wake 2005; Hansen and Wake, pers. comm. 2008; CaliforniaHerps 2011; Evelyn, pers. comm. 2012

Note: Land cover types are a necessary but not sufficient condition for occurrence. The Tehachapi slender salamander is closely associated with talus and rocky slopes.

Foraging Requirements

Although the Tehachapi slender salamander's specific feeding habits are unknown, related species feed on small arthropods, such as spiders and mites, insects (especially collembolans, coleopterans, and hymenopterans), earthworms, and snails (Cunningham 1960; Adams 1968). The Tehachapi slender salamander primarily forages under surface objects, such as pieces of bark or flat talus rocks, in moist areas or in leaf litter. *Batrachoseps* are generally sit-and-wait predators (CaliforniaHerps 2011); they search or wait for small insects and other invertebrates under surface objects (USFS 2006). Salamanders may enter termite tunnels and earthworm burrows when foraging (Morey 2005). It is assumed that the Tehachapi slender salamander, similar to all *Batrachoseps* species observed thus far, capture small invertebrates using a projectile tongue (Hansen and Wake 2005). As a semifossorial species, the Tehachapi slender salamander is able to enter termite tunnels, earthworm burrows, and other small openings not accessible to larger salamanders. They may compete with juvenile salamanders of other species where their ranges overlap (Morey 2005).

Reproduction

Reproduction by *Batrachoseps* species is terrestrial (Hansen and Wake 2005). Eggs are laid in moist places under surface objects and neonates hatch fully formed (USFS 2006; CaliforniaHerps 2011). The breeding season of the Tehachapi slender salamander is suspected to be from about November to February, with peak activity in November and December, but the timing of reproduction is likely climate related (see Table 2). The Tehachapi slender salamander probably lays eggs during the rainy periods of winter and early spring (Morey 2005). Breeding activity may extend into May at higher elevation and at sites with moist conditions. Clutch size remains unknown, although related salamanders lay eggs in clusters of 4 to 21 (Stebbins 1954; USFS 2006).

Although nest sites have not been directly observed, eggs are likely deposited deep within the rock talus and litter matrix typical of Tehachapi slender salamander microhabitat (Hansen and Wake 2005). Tehachapi slender salamanders may build communal nests,

which have been reported for the sympatric black-bellied salamander (Jockusch and Mahoney 1997).

Table 2. Key Seasonal Periods for Tehachapi Slender Salamander

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding	X	X	X	X	X						X	X
Aestivation					X	X	X	X	X	X		

Notes: Surface and breeding activity likely is associated with precipitation and may extend into May with high precipitation and at higher elevations and sites with moist conditions. During dry years or extended periods of drought, salamanders may remain below the surface.
Sources: Hansen and Wake 2005; Morey 2005.

Spatial Activity

The Tehachapi slender salamander is not thought to be territorial (USFS 2006), although females of related species are often found in the immediate vicinity of egg clusters (Morey 2005). Tehachapi slender salamander home ranges are suspected to be approximately 0.5 acre (USFS 2006), with individuals moving no more than about 164 feet in their lifetime (Hansen and Wake, pers. comm. 2008). The area of Tehachapi slender salamander surface activity probably covers its area of underground activity (Morey 2005). In similar slender salamander species, up to 15 individual territories have been located within a 1,076-square-foot area (Hansen and Wake, pers. comm. 2008).

The activity patterns of the Tehachapi slender salamander are largely dependent upon precipitation patterns, which are erratic in both timing and amount within the species' range (Hansen and Wake 2005). Surface activity closely relates to the onset of the rainy season, which generally occurs around November or December (Hansen and Wake 2005). At lower elevations this rainy season may be rather brief (2 to 3 months) (Hansen and Wake 2005). Due to the relative dryness of its habitat, the Tehachapi slender salamander may have a shorter activity period than other slender salamanders (CaliforniaHerps 2011). During the moist period (November to May) the Tehachapi slender salamander can be found nocturnally active on the surface,

although periods of surface activity vary from year to year (Morey 2005). March and April generally marks the salamander's peak surface activity, although it can extend into May in wet years or at higher elevations (e.g., upper reaches of Pastoria and Tejon Creek drainages, Tehachapi Mountains) (Hanson and Wake, pers. comm. 2008). During drier periods, salamanders retreat underground to moist seepages (Morey 2005). In years of below-average rainfall or consecutive years of drought, salamanders may not appear under surface cover at all, but rather retreat to subterranean refugia (Morey 2005; Hansen and Wake 2005). The portion of the species' range in the Plan Area is lower elevation and drier than the more westerly and higher elevation portions of its range, and it is expected to spend more time underground in this part of its range.

Ecological Relationships

All known Tehachapi slender salamander localities overlap the range of the yellow-blotched salamander (*Ensatina eschscholtzii croceater*) (Hansen and Wake 2005). Both species occupy similar habitats, but yellow-blotched salamanders have a more extensive distribution. In some areas where yellow-blotched salamanders are abundant, Tehachapi slender salamanders do not occur; conversely, where Tehachapi slender salamanders are locally abundant there are few yellow-blotched salamanders. Tehachapi slender salamanders and yellow-blotched salamanders are the only salamanders present in Caliente Canyon, although black-bellied slender salamanders and possibly gregarious slender salamanders (*Batrachoseps gregarious*) are believed to occur nearby (Hansen and Wake 2005). Within the Tehachapi Mountains, Tehachapi slender salamanders and black-bellied slender salamanders are sympatric in the Pastoria and Tejon Creek drainages, at Fort Tejon in Grapevine Canyon, and possibly elsewhere (Jockusch 1996; Wake and Jockusch 2000) but do not hybridize (Hansen and Wake, pers. comm. 2008). Tehachapi slender salamanders are habitat specialists, whereas black-bellied slender salamanders occupy a broader distribution. The sympatric relationship between these two species is notable given that it is the only case of sympatry involving members of the same species group of *Batrachoseps* (Wake and Jockusch 2000).

Primary predators of the Tehachapi slender salamander are most likely small snakes such as the ring-necked snake (*Diadophis punctatus*) (Hansen and Wake 2005). Other potential predators of both adults and juveniles include beetle larvae and other predatory arthropods, diurnal birds (especially birds that forage through leaf litter), and small mammals (Morey 2005).

Population Status and Trends

Global: Imperiled (NatureServe 2010)
State: Same as above
Within Plan Area: Unknown

Population trends of the Tehachapi slender salamander are unknown. However, all documented occurrences are considered to be extant, although individual populations are small and localized (Hammerson 2009). No ecological or population studies have been conducted that would provide specific information about population status and trends.

Threats and Environmental Stressors

Tehachapi slender salamander populations are restricted to seasonally shaded, north-facing slopes of canyons located in otherwise arid to semi-arid terrain where talus occurs. The small and localized nature of these populations, which occur at a limited number of sites, makes them highly susceptible to habitat disturbance caused by development. The USFWS analyzed the threat to Tehachapi slender salamander posed by proposed development in the 12-month finding (76 FR 62900–62926). The only known potential development-related threats to the species are the proposed Tejon Mountain Village residential and commercial development in the Tehachapi Mountains. The USFWS found that under a worst-case scenario only 2.8% of suitable habitat for the species would be impacted by the Tejon Mountain Village development and concluded that this level of impact would not threaten the Tehachapi Mountains DPS (76 FR 62900–62926).

Within the Plan Area, identified threats at two of the recent (2007, 2009) documented sites include possible erosion from the paved road at the site south of Caliente Creek Road (CDFW 2013). The CNDDB (CDFW 2013) indicates that the area of the Tollgate Canyon/Stevenson

Creek site is proposed for wind energy development (CDFW 2013). However, the USFWS 12-month finding does not identify wind energy development as a potential threat at this site (76 FR 62900–62926). The sites at Silver Creek, Indian Creek, and the unnamed canyon south of Indian Creek are on private lands. Based on site photographs, the Silver Creek and Indian Creek sites appear to be in fair to good condition because grazing occurs at the sites, but there are no signs of other activities, such as buildings, roads, or mining (76 FR 62900–62926). The site at the unnamed canyon south of Indian Creek appears to be in good condition based on site photographs. This site is on BLM land and there is no evidence of grazing, nor is it within a BLM grazing allotment (76 FR 62900–62926). No other threats were identified for these new sites.

Tehachapi slender salamander habitat is also potentially threatened by feral pig (*Sus scrofa*) (Hansen and Wake, pers. comm. 2008), road construction, mining, and cattle grazing, as well as flood control projects (Hansen and Stafford 1994; Jennings 1996). Hansen and Wake (pers. comm. 2008) considered feral pigs to be the main threat to Tehachapi slender salamander in the Tehachapi Mountains.

The USFWS analyzed the potential effects of climate change on the Tehachapi slender salamander in the 12-month finding (76 FR 62900–62926). Based on the climate models, temperatures in the Tehachapi Mountains are expected to increase, but the effect of climate change on precipitation is less certain. There is a high level uncertainty as to how these changes will affect Tehachapi slender salamander (76 FR 62900–62926). While any specific effects on the species remains speculative, the USFWS concluded that some loss of habitat may occur in more exposed canyon areas, but that habitat will remain in the most shaded, lower portions of the canyons and that the species may also be able to shift within canyons in response to climate change (76 FR 62900–62926).

Conservation and Management Activities

Three of the five recent occurrences in the Plan Area are on BLM land. (the 2007 and 2009 occurrences and the 2011 occurrence in the unnamed canyon south of Indian Creek). BLM Manual 6840 establishes Special-Status Species policy for plant and animal species

and the habitat on which they depend (BLM 2001). The objectives of the BLM policy are:

A. To conserve listed species and the ecosystems on which they depend.

B. To ensure that actions requiring authorization or approval by the BLM are consistent with the conservation needs of special status species and do not contribute to the need to list any special status species, either under provisions of the ESA or other provisions of this policy (BLM 2001).

The BLM has identified the Tehachapi slender salamander as a sensitive species and requires surveys in suitable habitat areas prior to authorizing activities that could impact the species or its habitat. However, because the species is not federally listed, the BLM is not legally required to avoid or mitigate agency-related impacts (74 FR 18336–18431).

The Tehachapi Upland Multiple Species Habitat Conservation Plan (TU MSHCP) would cover occupied Tehachapi slender salamander habitat west of the Plan Area. The TU MSHCP, currently under review by the USFWS, would conserve and manage approximately 3,507 acres (95%) of modeled suitable habitat for the species on Tejon Ranch, and all currently documented locations of Tehachapi slender salamander (Monroe and Bear Trap canyons) would be protected in open space. The protection and management of modeled suitable habitat for the Tehachapi slender salamander would occur in the context of the much larger open space system planned for Tejon Ranch, which would ultimately preserve up to 240,000 acres of the 270,000-acre ranch per the Tejon Ranch Conservation and Land Use Agreement (TRC et al. 2008). As currently proposed, the TU MSHCP (Dudek 2009) would also implement avoidance, minimization, and mitigation measures for the species and its modeled suitable habitat during development and long-term operation of the Tejon Mountain Village Project, including:

- Avoidance of ground disturbances in modeled suitable habitat except as necessary for road crossing and culverts
- Implementation of best management practices (BMPs) to protect surface water quality

- Pre-construction surveys and relocation of detected individuals to suitable habitat outside construction areas, and biological monitoring during all ground-disturbing activities within modeled suitable habitat areas
- Design features between development and modeled suitable habitat to avoid and minimize adverse edge effects, such as exotic plant and animal species (e.g., Argentine ant [*Linepithema humile*]) and controls on lighting adjacent to open space
- Implementation of a grazing management plan to maintain habitat for the species
- Homeowner education and controls on recreational activities and pets
- Environmental baseline surveys
- Minimization of infrastructure impacts in open space and use of BMPs for the design and installation of such infrastructure
- Selection of appropriate locations for public access, trails, and facilities to minimize impacts to open space areas.

Data Characterization

Little occurrence data are available for the Tehachapi slender salamander, and the special details of its life history are largely unknown (Hansen and Wake 2005). As discussed previously, there are 16 occurrence records for the species in the CNDDB (CDFW 2013) and three very recent occurrence records included in the USFWS 12-month finding (76 FR 62900–62926). Much of the potential habitat area is on private lands and not readily accessible to biologists (Hansen and Wake 2005). However, even when broad-scale focused surveys are conducted for the species, detections are few. Detection of this species is difficult, even where it is present. Surveys need to take place during the right time of year when conditions are appropriate and be carried out by people with experience finding the species (Evelyn, pers. comm. 2012). For example, focused surveys for the species were conducted within the approximately 26,400-acre Tejon Mountain Village project area in 2007. Focused surveys were conducted in 60 drainages considered to support suitable habitat for the species, but it was documented in only one of the 60 drainages (i.e., Monroe Canyon) (Jones & Stokes 2008).

Within this survey area, there are only four other documented occurrences in the CNDDB (CDFW 2013). This species is only active on the surface for a limited time period during the wet season and spends most of its life underground. Detecting individuals on a large scale would require unacceptable and destructive survey methods (e.g., excavations and turning up rocks and other materials), although it is feasible that some type of systematic or random sampling regime to minimize habitat damage could be used. To date, no such sampling regime has been implemented.

As described previously, there are six occurrence records for the species in the Plan Area, including one historic record and five recent records (note that the Big Last Chance Canyon record is considered recent because the species was last detected there in 2007; it was first reported by Brame and Murray (1968)).

Management and Monitoring Considerations

As described previously, BLM Manual 6840 provides policy direction for management of sensitive species, including Tehachapi slender salamander (BLM 2001). The BLM policy is to use the best available scientific information for adequate review of a land-use plan or other proposed agency action. This may include baseline studies, management, and monitoring of management actions. Management should consider potential ongoing threats, such as livestock grazing, which can degrade the woodland and riparian habitats occupied by the Tehachapi slender salamander, including vegetation structure, soils, microhabitat (e.g., talus and rocks), and water quality. Other considerations for management and monitoring include potential adverse edge effects in suitable habitat, such as erosion and polluted runoff into habitat areas, including pesticides and other chemicals. Because this species breathes through its highly permeable skin, it is likely highly vulnerable to environmental toxins and dust mediated through the air and water that can be absorbed through the skin. Lighting can make this nocturnal species more visible to predators. Invasive plant and animal species can degrade habitat, displace native species, and result in increased predation (e.g., pet and feral cats). Development or other land uses that facilitate both authorized and unauthorized public access to occupied areas can result in habitat

degradation (e.g., disturbance of talus slopes and drainages) and impacts to individuals (e.g., illegal collecting). For wind energy projects, for example, when siting turbines, new access roads need to be considered in addition to the actual turbine footprint. In steep terrain, road construction could decimate a hillside and potentially disrupt multiple discrete populations of Tehachapi slender salamander. Therefore, prior to road design, potential routes should thoroughly and surveyed for potential Tehachapi slender salamander habitat (Evelyn, pers. comm. 2012).

Management also should focus on maintaining existing habitat connectivity among occupied areas to the extent feasible. Because this species is likely very sedentary (Hansen and Wake 2005), it is probably not capable of dispersing long distances through unsuitable or marginal habitat. It is likely that local populations are already naturally isolated by unsuitable habitat. Development and land uses that are incompatible with occupation may fragment habitat and further isolate small populations, potentially leaving them vulnerable to local extinction due to lack of gene flow, inbreeding depression, reduced genetic diversity, and genetic drift.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Tehachapi slender salamander, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 47,883 acres of modeled suitable habitat for Tehachapi slender salamander in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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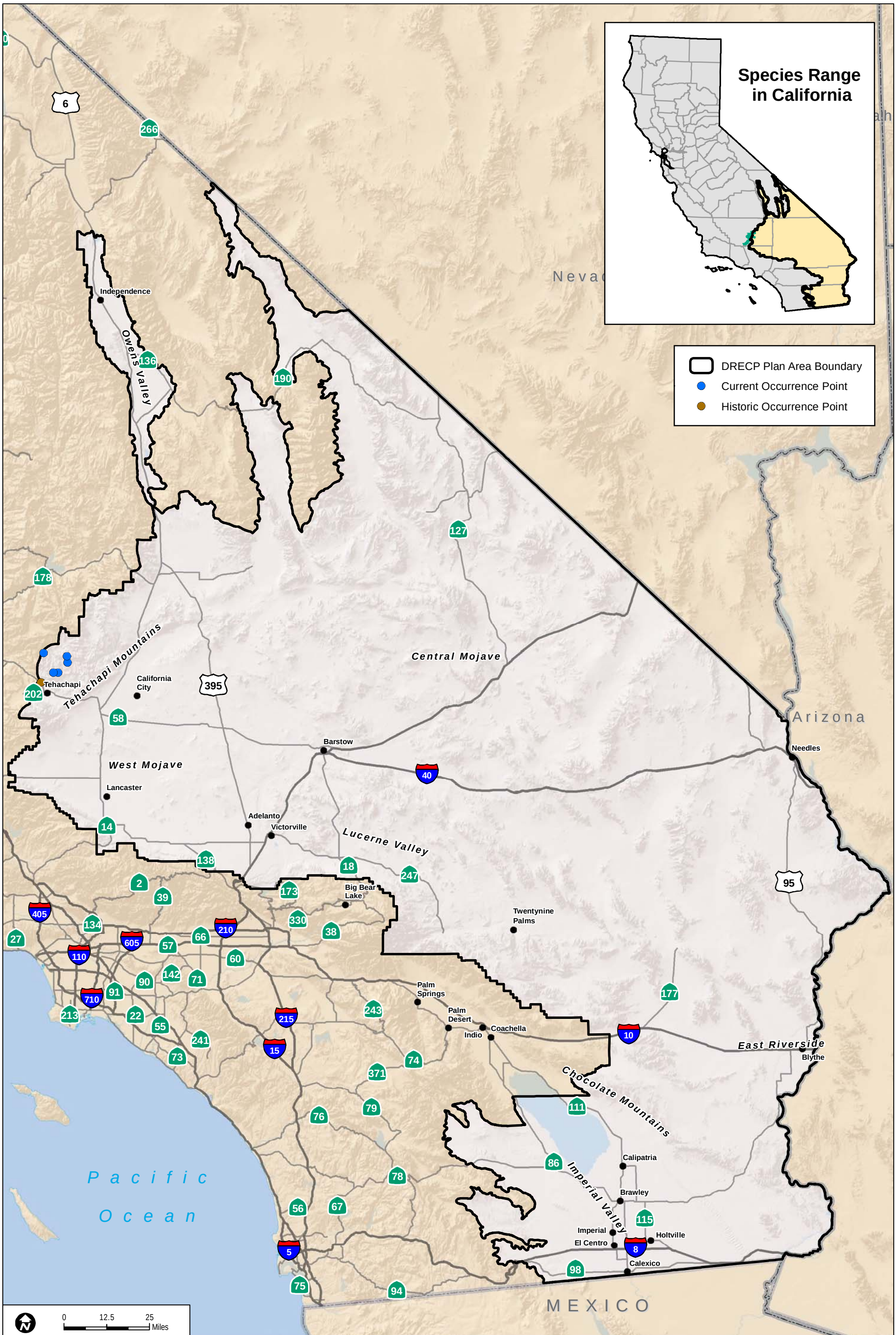

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-A01

Tehachapi Slender Salamander Occurrences in the Plan Area

Bendire's Thrasher (*Toxostoma bendirei*)

Photo courtesy of Stephen Dowlan.

Legal Status

State: Species of Special Concern
Federal: Bureau of Land Management Sensitive; U.S. Fish and Wildlife Service Bird of Conservation Concern
Critical Habitat: N/A
Recovery Planning: N/A
Notes: IUCN Conservation Status: Vulnerable (BirdLife International 2012) and on the American Bird Conservancy U.S. WatchList of Birds of Conservation Concern (CDFG 2011).

Taxonomy

Bendire's thrasher (*Toxostoma bendirei*) was first collected and described by Major Charles E. Bendire in 1872 near current downtown Tucson, Arizona. At the time of its first description, Robert Ridgeway believed it to be a female of another species (Curve-billed thrasher, *T. curvirostre*) and Elliot Coues was hesitant on its taxonomy (Coues 1873).

Rossem (1942) described two additional races of Bendire's thrasher occurring in Sonora based on their coloration. Based on these descriptions, Miller et al. (1957) and Mayr and Greenway (1960) recognize three subspecies: *T. b. bendirei*, *T. b. candidum*, and *T. b. rubricatum*. However, these subspecies are not recognized by the American Ornithologists' Union (1998), Unitt (2004), and Phillips (1986)and Phillips (1986) states that the differences in appearance of *T. b. candidum* and *T. b. rubricatum* are those due to season, wear, and fading.

Bendire's thrasher is considered a member of the curve-billed thrasher complex which includes the curve-billed thrasher, ocellated thrasher (*T. ocellatum*), and gray thrasher (*T. cinerium*) (England and Laudenslayer Jr. 1993). It has been proposed that isolation during glacial periods resulted in the differentiation among the members of the complex of species (England and Laudenslayer Jr. 1993).

Physical characteristics of the species are detailed by England and Laudenslayer Jr. (1993).

Distribution

General

The exact distribution of this species is poorly understood due to its secretive behavior, migratory movements, and lack of research (England and Laudenslayer Jr. 1993). In general, this species is found in the southwestern U.S. deserts ranging from southeastern California, southernmost Nevada, southernmost Utah, southern Colorado south through New Mexico, and throughout the Sonora desert. In Mexico, the species distribution is believed to be in Sonora with wintering to Tiburon Island and northern Sinaloa (Blake 1953). The species appears to be mostly confined to the Mojave Desert (Unitt 2004), and northwestern Mexico deserts (England and Laudenslayer Jr. 1993).

Bendire's thrasher is known to breed from southeastern California, southern Nevada, southern Utah, south-central Colorado, western and throughout New Mexico (Darling 1970), south to central Sonora, and throughout Arizona (Miller et al. 1957; Phillips et al. 1964; England and Laudenslayer Jr. 1989a, 1989b; AOU 1998). Within New Mexico and California, breeding appears irregular leaving many suitable sites unoccupied (England and Laudenslayer Jr. 1993).

In winter, Bendire's thrasher leaves the northern areas of its breeding range (England and Laudenslayer Jr. 1993). Bendire's thrashers that breed in California are thought to winter in southern Arizona, southwestern New Mexico, and Sonora, Mexico (England and Laudenslayer Jr. 1989a, 1989b). One record also exists for the species detection as far south as southern Sinaloa, Mexico (Bent 1948).

Distribution and Occurrences within the Plan Area

Historical

Overall, there are approximately 62 historical (i.e., pre-1990) Bendire's thrasher occurrence records in the Plan Area (CDFW 2013; Dudek 2013). These occurrences are located in eastern Kern County, throughout San Bernardino County, and central Riverside County (Figure SP-B03) with the majority of occurrences detected in San Bernardino County.

Within the Plan Area, most occurrences have historically occurred within or near the Mojave National Preserve and between Victorville and Joshua Tree National Park (Figure SP-B03) with approximately 38 records near or within the Mojave National Preserve in eastern San Bernardino (Figure SP-B03). Twenty-one additional records are documented between Victorville and south to Joshua Tree National Park. There are also three more disjunct records at the southern end of the Turtle Mountains, at the Naval Air Warfare Center China Lake, and south of Kern. Historically, this species was considered to breed primarily in the Mojave Desert (Grinnell and Miller 1944; Garrett and Dunn 1981), was considered common in summer in areas of northeastern San Bernardino County, and considered a sparse summer resident in the Joshua Tree National Monument-Yucca Valley area (McCaskie 1974; Remsen 1978).

Recent

Currently, there are approximately 11 recent (i.e., since 1990) Bendire's thrasher occurrences in the Plan Area in the following locations: Mojave National Preserve, east of Barstow, in and near Lucerne Valley, within or near Yucca Valley, near the junction of I-8 and SR-177, and near Lake Havasu City (CDFW 2013; Dudek 2013; Figure SP-B03).

In general, the species current distribution is similar to its historical distribution. Although plenty of undisturbed habitat exists, the reasons for the species rarity in California are not clear (Unitt 2004). It has been estimated that the population may be fewer than 200 pairs throughout California (Remsen 1978). However, the exact distribution and population status of this species is unknown.

Natural History

Habitat Requirements

Bendire's thrashers typically breed in open grasslands, shrubland, or woodland with scattered trees and shrubs (England and Laudenslayer Jr. 1993). The vegetation within occupied areas may vary depending on the elevation which ranges from 0 to 5,900 feet (England and Laudenslayer Jr. 1993). At high elevations the species may be associated with sagebrush

(*Artemisia* sp.) and some junipers (*Juniperus* sp.). At lower elevations it is associated with deserts and grasslands, such as the Mojave desert scrub. Characteristic plant species within areas where it occurs include Joshua trees (*Yucca brevifolia*), Spanish Bayonet (*Y. baccata*), Mojave Yucca (*Y. schidigera*), cholla cactus (*Opuntia* spp.) and/or other succulents, palo verde (*Cercidium* spp.), mesquite (*Prosopis* spp.), catclaw (*Acacia* spp.), desert-thorn (*Lycium* spp.), and agave (*Agave* spp.) (England and Laudenslayer Jr. 1989a, 1989b, 1993).

Bendire's thrashers may occasionally use vegetation around human habitation and agriculture when the habitat structure resembles natural habitat and curve-billed thrashers are absent (Gilman 1915a, Phillips et al. 1964, Rosenberg et al. 1991).

Little information exists for specific habitats used in migration or on wintering grounds, although wintering habitat plant community structure is similar to that used during the breeding season (England and Laudenslayer Jr. 1993).

Table 1. Habitat Associations for Bendire's Thrasher

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Data
Desert scrub	Breeding, foraging	Primary	Typically breeds in open grasslands, shrubland, or woodland with scattered trees and shrubs	England and Laudenslayer Jr. 1993

Foraging Requirements

Bendire's thrashers mainly consume insects and other arthropods; however, they may also consume seeds and berries (Ambrose Jr. 1963). The only quantitative study on the stomach contents of this species found ants, termites, and Lepidoptera larvae to dominate (Ambrose 1963). Anecdotal reports of birds foraging or carrying prey to the nest suggest that grasshoppers, beetles, caterpillars, and other larvae or pupae that it obtains near or on the ground dominate the diet (Woodbury 1939, Engels 1940, Bent 1948).

Typically, Bendire's thrashers forage on the ground but may also search vegetation for insects and pick fruit (Engels 1940; Ambrose 1963). This species uses its bill to peck, probe, and hammer in the ground (Engels 1940). They may occasionally use their bill to dig, but may not be efficient in this use (Ambrose 1963). They are not known to scratch the ground with their feet (Ambrose 1963).

Reproduction

In California, territorial behavior begins when the species returns to the breeding grounds beginning in mid-March through mid-June (England and Laudenslayer Jr. 1989a, 1989b). In Arizona, this species may return to breeding sites in small unmated flocks as early as the beginning of February (earliest date February 9; see Brown 1901). There is no additional information on how pair formation begins, where it occurs, or the process of nest construction in this species (England and Laudenslayer Jr. 1993).

Nests have been reported with eggs in early March (Arizona; Brown 1901) and late March (California; England and Laudenslayer Jr. 1993) suggesting nest building begins shortly after arriving to the breeding grounds. Clutches are typically 3-4 eggs (Brown 1901). Historical data reviewed by England and Laudenslayer Jr. (1993) suggest, although is not definitive, the breeding begins earlier in the southeast and advances across to the northwest of their breeding range.

Bendire's thrashers have been known to produce a second clutch in a season (England and Laudenslayer Jr. 1989a, 1989b). Only one record exists for the occurrence of a third brood in a season (Gilman 1915a).

Bendire's thrashers typically breed in dry scrub and cacti of desert areas. Nests may be low in a tree, shrub, or cactus clumps and usually 2 to 4 feet off the ground; occasionally 12 feet high (Baicich and Harrison 1997). The most common nest host plants include cholla, juniper, mesquite, Joshua trees and other yuccas (England and Laudenslayer Jr. 1993; Darling 1970).

Table 2. Key Seasonal Periods for Bendire's Thrasher

	Jan	Feb	Mar	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding			✓	✓	✓	✓	✓					
Migration		✓	✓	✓	✓	✓	✓	✓	✓	✓		
Wintering	✓									✓	✓	✓

Sources: England and Laudenslayer Jr. 1989a, 1989b, 1993, see Figure 4.

Spatial Behavior

There is no information on the specific territoriality behavior of this species. Overall, this species is migratory in the northern portion of their range and a permanent resident in the southern portion. In the northern portion of their range, dispersal may begin directly after breeding (England and Laudenslayer Jr. 1993).

Ecological Relationships

There is one record of a Bendire's thrasher nest being parasitized by a brown-headed cowbird (*Molothrus ater*) (three Bendire's thrasher eggs with one cowbird; Friedman 1934).

Information does not exist for the level of predation on this species. However, there is one record for a Gila woodpecker (*Melanerpes uropygialis*) pouncing on a Bendire's thrasher that successfully escaped (Gilman 1915b). Gilman (1915b) has observed Gila woodpeckers beginning to attack Bendire's thrashers.

Young in post-breeding flocks have been observed to be mixed with a few curve-billed and Crissal thrashers (*T. crissale*) (Scott 1888). In general, Bendire's thrashers may be observed in pairs or immediately after breeding in small flocks. However, they are usually inconspicuous except when singing (England and Laudenslayer Jr. 1993).

Ambrose (1963) suggests that possible competition with curve-billed thrashers for an exhausted food supply was contributing to the population decline. Curve-billed thrashers are sympatric throughout parts of this species range (Tweit 1996; Engels 1940; Ambrose 1963; Tomoff 1974).

Population Status and Trends

Global: Suspected decline; however, trends are poorly documented (BirdLife International 2013). Population estimated to be 170,000 (Audubon 2013).
State: Not clear
Within Study Area: Not clear

Information is lacking on the exact population status and trends of Bendire's thrashers. Unfortunately, population trends cannot be reliably estimated for this species from the North American Breeding Bird Survey (see *Regional Credibility* in Sauer et al. 2008). Records from the Breeding Bird Survey counts (from Arizona, California, Colorado, Nevada, New Mexico, and Utah) are infrequent for this species, and no significant trends could be detected for the period from 1965 to 1979 (Robbins et al. 1986; England and Laudenslayer Jr. 1993).

Declines over 37 years (1966–2003) are estimated at 34.5% (BirdLife International 2013). It is suggested that population may have declined in areas of Arizona between 1940 and 1960 (Ambrose 1963). Unfortunately, the historical and most current field investigations (England and Laudenslayer Jr. 1989a, 1989b) were inadequate to determine the population status or trends of the species in California.

Remsen (1978) suggested the total California population was under 200 pairs. Due to these concerns, the species was listed on the California Department of Fish and Game Birds Species of Special Concern (Remsen 1978). As such, there is concern for the status of this species due to their disjunct distribution, seemingly isolated populations, and unknown population sizes. However, in New Mexico, one report suggests the range of the species may have expanded into areas with junipers due to overgrazing (Darling 1970). Populations around Tucson may have been reduced by urbanization (density of 0.2 birds/100 acres in desert areas and none in urban; Emlen 1974) and agricultural efforts near the Gila River (Rea 1983).

Threats and Environmental Stressors

Although more research needs to be conducted, Remsen (1978) suggests the Bendire's thrasher is threatened by habitat destruction/alteration

(specifically with the harvesting of Joshua trees and yucca), overgrazing, and off-road vehicle use in their breeding habitats. This species may also be threatened by loss of breeding habitat to urban and agricultural development as well as military operations (Shuford and Gardali 2008). However, without any existing quantitative information regarding population densities, most of the information on threats comes from anecdotal descriptions of the species (England and Laudenslayer Jr. 1989a, 1989b).

Ambrose (1963) suggests that possible competition with curve-billed thrashers for an exhausted food supply was contributing to the population's decline. Curve-billed thrashers are sympatric throughout parts of this species range (Tweit 1996; Engels 1940; Ambrose 1963; Tomoff 1974). However, Engels (1940) suggested that the means of ecological separation of these species cannot be concluded.

Anecdotal reports suggest that populations may persist in agricultural areas bordered by mesquite and other shrubs (Ambrose 1963) as well as in rural areas with dwellings near vegetation (Gilman 1915a; Rea 1983).

Conservation and Management Activities

There is no information on other management actions for any states in this species range (England and Laudenslayer Jr. 1993).

Data Characterization

In general, there is a lack of information of Bendire's thrashers throughout their range.

Management and Monitoring Considerations

England and Laudenslayer Jr. (1989b) concluded that (1) the breeding population of Bendire's thrashers was more widely distributed than previously documented, and (2) there is inadequate understanding of this species ecology and population. They recommended several long-term research and population monitoring considerations:

1. Conduct long-term (10+ years) monitoring of isolated populations throughout the Mojave Desert.

2. Survey habitat that appears suitable but lacking breeding records to locate additional breeding populations.
3. Survey the Colorado Desert to identify breeding locations and habitats use; current data suggest possible regular breeding in small numbers.
4. Examine the species breeding biology (e.g., reproductive phenology, food habits, nesting ecology, foraging habits) in order to build a basic understanding of the species that may inform future management recommendations.
5. Examine the impact of desert land use on this species (e.g., urbanization, grazing, off-road vehicle use, removal of select vegetation species). The results of these efforts may also inform management on other species impacted by desert land use.

Shuford and Gardali (2008) also suggest the following monitoring: (a) examine possible competition between northern mockingbirds (*Mimus polyglottus*) and Bendire's thrashers to determine their effect on the species, (b) create conservation management areas for the species on public (BLM) lands, (c) examine factors influencing the species reproductive success and annual survivorship, and (d) identify areas that serve as population sources and sinks.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Bendire's thrasher, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

The model generated 2,216,932 acres of modeled suitable habitat for Bendire's thrasher in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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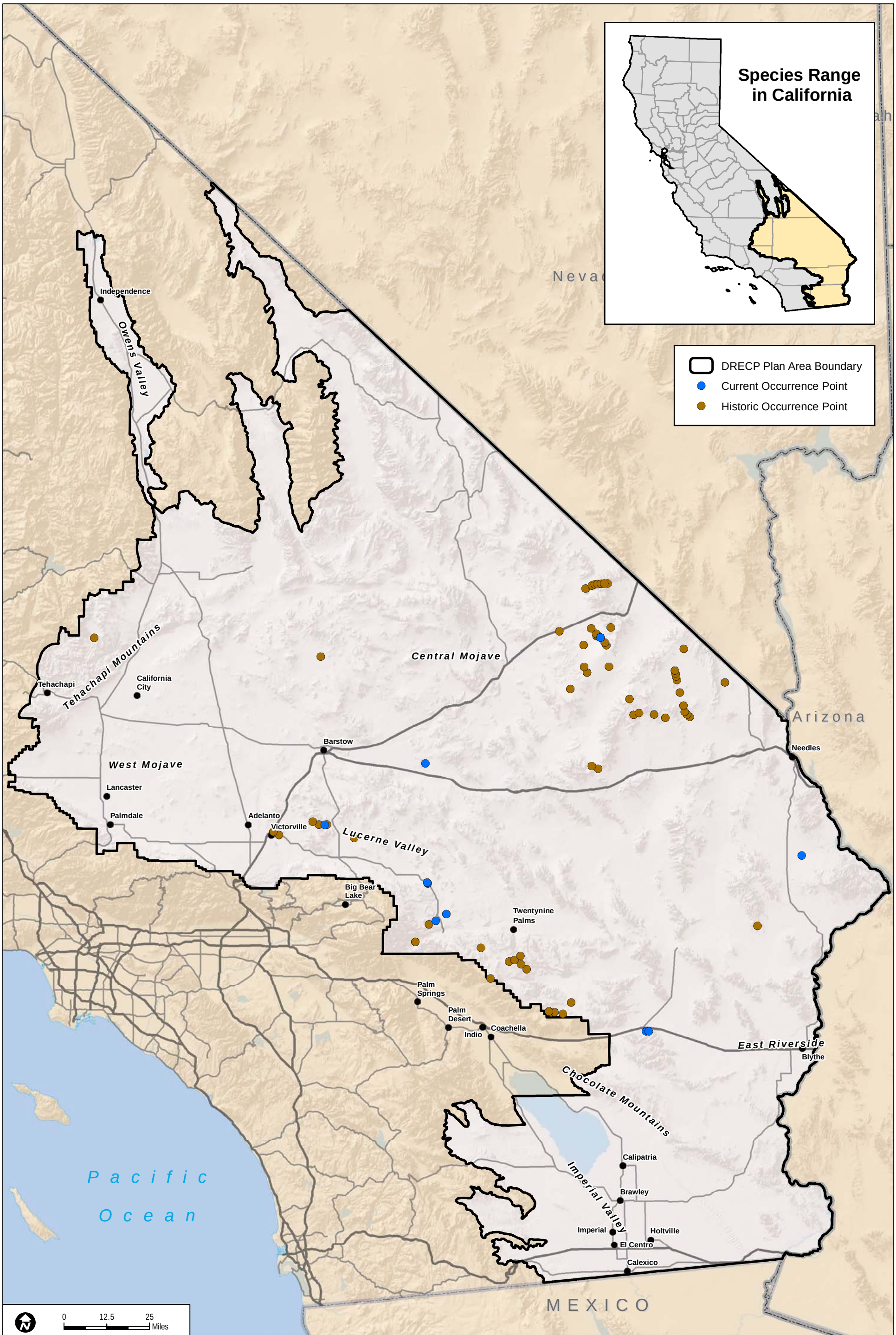

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B02

Bendire's Thrasher Occurrences in the Plan Area

Burrowing Owl (*Athene cunicularia* ssp. *hypugaea*)

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Legal Status

State of California: Species of Special Concern
Federal: Bureau of Land Management Sensitive, U.S. Fish and Wildlife Service Bird | of Conservation Concern
Other: Endangered in Canada and Minnesota; Threatened in Colorado; Mexico: "Special Protection" status.
Critical Habitat: N/A
Recovery Planning: N/A
Notes: The burrowing owl has been included on the list of California Species of Special Concern since 1978 (Remsen 1978; Gervais et al. 2008). In 2003, a petition to list the burrowing owl as threatened or endangered under the California Endangered Species Act (Center for Biological Diversity et al. 2003) was rejected by the California Fish and Game Commission (Miller 2007). Populations in California continue to decline or have been extirpated from rapid loss of farmland, changes in agricultural practices, eradication of ground squirrels, pesticide use, traffic and wind turbine-related mortality, and possibly West Nile virus (Gervais et al. 2008). Another petition could be submitted, however, that could potentially change the burrowing owl's status during the planning and implementation of the DRECP.

Taxonomy

Up to 25 subspecies have been recognized (Poulin et al. 2011), but only one subspecies (*Athene cunicularia hypugaea*) occurs in continental North America outside of Florida (Poulin et al. 2011). Descriptions of the species' physical characteristics, behavior, and distribution are provided in the most recent revision of the species

account for Birds of North America revised by R. Poulin and L.D. Todd (Poulin et al. 2011).

Distribution

General

Western burrowing owl is found in non-mountainous western North America, from the Great Plains grasslands in southern portions of the western Canadian provinces south through the U.S. into Mexico (Poulin et al. 2011). Other subspecies occur in arid, open habitats in Florida, the Caribbean Basin, and South America (Poulin et al. 2011; Clark 1997) (Figure SP-B04).

In California, the burrowing owl's range extends throughout the lowlands from the northern Central Valley to the U.S./Mexico border, with about two-thirds of the population occupying the Imperial Valley, near the Salton Sea (Wilkerson and Siegel 2010). The species' distribution and abundance vary considerably throughout its range (DeSante et al. 2007; Wilkerson and Siegel 2010). Breeding burrowing owls are generally absent from the coast north of Sonoma County and from high mountain areas, such as the Sierra Nevada and the Transverse Ranges extending east from Santa Barbara County to San Bernardino County (Gervais et al. 2008).

Distribution and Occurrences within the Plan Area

Historical

Grinnell and Miller (1944) described a range in California that included most of the lowlands, although "mostly rare or wanting in coastal counties north of Marin County" with "Numbers in favorable localities large; originally common, even 'abundant'." They regarded the species as "becoming scarce in settled parts of the State" due to "roadside shooting, anti-'vermin' campaigns, elimination of ground squirrels—hence of nesting places for these owls." The increase in abundance of burrowing owls in some agricultural environments, such as the Imperial Valley, likely began when the native desert ecosystem in this region was converted to large areas of irrigated agriculture (DeSante et al. 2004). The time period for this shift was in

the early 20th century as van Rossem (1911) considered the species "abundant everywhere in suitable locations" in the Imperial Valley.

Recent

The overall range of the burrowing owl in California has not drastically changed from that described by Grinnell and Miller (1944), but the species has disappeared or greatly declined as a breeding bird in many areas that were once occupied (DeSante et al. 2007; Gervais et al. 2008; Wilkerson and Siegel 2010). By one recent estimate (Miller 2007), the burrowing owl has functionally disappeared as a breeding species from 22% of its former range and continues to decline in an additional 23% of its range.

A statewide survey conducted from 1991 to 1993 found that populations had disappeared from the central coast (Marin, San Francisco, Santa Cruz, Napa, and coastal San Luis Obispo counties), Ventura County, and the Coachella Valley in Riverside County, and were nearly extirpated from Sonoma, Santa Barbara, Orange, coastal Monterey, and San Mateo counties, where only small, remnant populations remained (DeSante et al. 2007).

The most current information on the burrowing owl's breeding distribution in California comes from systematic surveys conducted in 2006-2007 across the species' mainland breeding range in the state (Wilkerson and Siegel 2010). Compared with the surveys in the early 1990s, this survey found 10.9% fewer pairs, but the overall change was not statistically significant. About 69% of California's population was found to be concentrated in agricultural areas of the Imperial Valley; secondary centers of abundance were identified in the southern Central Valley (~12% of the state total), middle Central Valley (~6% of the state total), western Mojave Desert (~6% of the state total), and Palo Verde Valley near Blythe in eastern Riverside County (~2% of the state total); approximately 5% of the state's population was scattered elsewhere.

Natural History

Habitat Requirements

Throughout their range, western burrowing owls require habitats with three basic attributes: open, well-drained terrain; short, sparse vegetation generally lacking trees; and underground burrows or burrow-like structures (e.g., culverts) (Klute et al. 2003; Gervais et al. 2008). Burrowing owls occupy grasslands, deserts, sagebrush scrub, agricultural areas (including pastures and untilled margins of cropland), earthen levees and berms, a variety of habitat types on coastal uplands (especially by over-wintering migrants) (California Natural Diversity Database 2010), and urban vacant lots, as well as the margins of airports, golf courses, residential developments, and roads (CVAG et al. 2007; Gervais et al. 2008). Burrowing owls occur on relatively flat expanses with level to gentle topography (CDFG 2012).

Several habitat characteristics may explain the species' distribution within the Plan Area: vegetation density, availability of suitable prey, availability of burrows or suitable soil, and disturbance (primarily from humans) (BLM 2005). However, Unitt (2004) notes that sites with suitable characteristics for burrowing owls may not support populations due to "high sensitivity to habitat fragmentation, proliferation of terrestrial predators, and high mortality from collisions with cars." During the breeding season, burrowing owls may need enough permanent cover and taller vegetation within their foraging range to provide them with sufficient prey, which includes large insects and small mammals (Poulin et al. 2011; Wellicome 1997). Paired males are known to line the burrow entrance and tunnel with dried mammal dung for several possible reasons including the prevention of nest predation and increasing insect presence near the nest as a source of convenient prey (Smith 2004). This behavior is obviously prominent in habitat that is regularly grazed by cows, horses or bison (Smith 2004).

Few desert areas have too much plant cover for burrowing owls; and those areas that do have high cover (e.g., palm oases) are unoccupied (e.g., Barrows 1989). Dense vegetation may not exclude burrowing owls directly, but rather indirectly through increased predation or competition with other species, or lowered hunting success for

preferred prey (BLM 2005). When vegetation height is greater than 5 centimeters (2 inches), owls may prefer habitat with elevated perches to increase their horizontal visibility to detect both predators and prey (Green and Anthony 1989). Suitable habitat associations for burrowing owl are summarized in Table 1.

Human alteration of the landscape can inadvertently or intentionally create suitable habitat, but can also make potential habitat unsuitable by way of "habitat loss, associated prey reduction, and human disturbance" (Lincer and Bloom 2007) and various pesticides are known to adversely affect burrowing owls, directly or indirectly (James and Fox 1987; Haug and Oliphant 1987). Agriculture and surface irrigation systems (i.e., earthen canals and ditches) can create habitat by providing bankside burrow sites and prey in the adjacent fields (Gervais et al. 2008; Poulin et al. 2011), while urban development and the associated excessive noise or disturbance can result in habitat loss and indirect adverse effects (BLM 2005).

Table 1. Habitat Associations for Burrowing Owl

Land Cover Type	Land Cover Use	Population Density	Habitat Parameters	Supporting Information
Shortgrass-dominated grasslands and steppes	Nesting, shelter, refugia	Medium	Burrows mostly dug by other animals including the California ground squirrel	The presence of nest burrows, dug by fossorial mammals such as ground squirrels, seems to be a critical requirement for burrowing owls. Typically forage in habitats characterized by low-growing vegetation (Poulin et al. 2011). Often use unlined earthen banks along agricultural ditches as burrow sites (Poulin et al. 2011)
Agricultural	Nesting, shelter, refugia	Varies, from low to the highest known.	See above	Rosenberg and Haley 2004; DeSante et al 2007.

Land Cover Type	Land Cover Use	Population Density	Habitat Parameters	Supporting Information
Desert Shrublands	Wintering range; less often, for breeding.	Extremely Low	See above	(Longshore and Crowe 2010; Wilkerson and Siegel 2011).
Urban-Suburban	Nesting, shelter, refugia	Low	See above	See above; may use urban levees if suitable burrows are available (Poulin et al. 2011)
Rural residential	Nesting, shelter, refugia	Low	See above	See above; may use urban levees if suitable burrows are available (Poulin et al. 2011)

Foraging Requirements

Burrowing owls are opportunistic predators that prey on arthropods, small mammals, birds, amphibians, and reptiles (Karalus and Eckert 1987; Poulin et al. 2011). Burrowing owls typically forage in habitats characterized by low-growing, sparse vegetation (Poulin et al. 2011) feeding on insects during the day, especially during the summer, and small mammals at night. Thomsen (1971) found that crickets and meadow voles (*Microtus* spp.) were the most common food items. Nocturnal foraging can occur up to several kilometers away from the burrow, and burrowing owls concentrate their hunting on grassland areas, crop fields, and structurally similar habitats with an abundance of small mammals (Haug and Oliphant 1990). The majority of the burrowing owl diet can be made up of rodents or large insects depending on the region in which they are found and the time of year (Rosenburg et al. 2007; Haug and Oliphant 1990).

Reproduction

Burrowing owls reach sexual maturity within one year of age (Poulin et al. 2011). Nesting in California generally runs from February through August, with peak activity from March to July (Zeiner et al. 1990; Thomsen 1971; Gervais et al. 2008).

Nesting sites always have available perching sites, such as fences or raised rodent mounds (Johnsgard 1988). Non-nest satellite burrows are typically employed to escape from approaching predators (especially raptors and ravens), to spread out pre-fledged nestlings (in case terrestrial predators invade one of an owl family's burrows and consume the young in it), and to relocate from parasite-infested nesting and roosting burrows (Dechant et al. 2002). Burrowing owls are primarily monogamous and typically breed once per year (Poulin et al. 2011). Mate fidelity between years was found to be high in the Imperial Valley (Catlin et al. 2005) but low in Saskatchewan (Poulin et al. 2011), perhaps reflecting a behavioral difference between resident and migratory populations. Normally, one clutch of 6–12 eggs is produced per year, with 7–9 eggs in a typical clutch (Poulin et al. 2011), although in rare instances two broods may be raised in a season (Gervais and Rosenberg 1999); the largest clutch recorded was 14 eggs, all of which hatched. Rosenberg et al. (2007) found variable productivity between habitat types, with productivity 10-20% lower in urban nest sites than grassland and fragmented habitat, but lowest in agricultural sites, which only average 2.9 ± 0.6 young per nest. Considerable variability also existed within years, where, even in an overall "good" or "poor" year, outlier nests existed. Clutch size is positively correlated with prey abundance (Wellicome 1997). Incubation normally lasts 28 to 30 days, beginning before the clutch is complete (Poulin et al. 2011). The eggs hatch asynchronously, which may be an adaptation to annual variation in prey abundance, whereby more young can be raised during years when prey is plentiful (Newton 1977, 1979; Wellicome 2005).

During incubation and brooding, the female stays in the burrow almost continuously while the male does the provisioning. Young burrowing owls fledge at about 44 days. As they mature they join the adults in foraging flights at dusk (Rosenberg et al. 1998). Prior studies in California have characterized burrowing owl reproductive success as 33% per nest attempt (Thomsen 1971) and 78% over seven breeding seasons (Trulio 1994, 1997), with 2.9 to 7.8 young fledged per successful nest (Poulin et al. 2011). However, burrowing owl fecundity in the Imperial Valley agricultural landscape is only 2.0 – 3.6 young fledged per nest (Rosenberg and Haley 2004).

Table 2. Key Seasonal Periods for Burrowing Owl

	Jan	Feb	Mar	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding			✓	✓	✓	✓	✓	✓				
Migration			✓	✓					✓	✓		
Winter Movements	✓										✓	✓

Source: Poulin et al. 2011

Spatial Behavior

Spatial activity includes migration by some individuals, dispersal, and home range use. Table 3 summarizes data for these activities.

California supports year-round resident burrowing owls and over-wintering migrants (Gervais et al. 2008). Many owls remain resident throughout the year in their breeding locales (especially in central and southern California) while some apparently migrate or disperse in the fall (Haug et al. 1993; Poulin et al. 2011; Coulombe 1971; Barclay 2007). Owls breeding in northern California locales and at higher elevations are believed to move south during the winter (Grinnell and Miller 1944; Haug et al. 1993; Zeiner et al. 1990). Other researchers report that burrowing owls may "wander" during the winter months, occasionally appearing and disappearing from their breeding grounds (McCaskie et al. 1988; Martin 1973).

It can be difficult to identify individual burrowing owls in mild-winter regions as being winter residents (migratory) seasonal wanderers, or permanent residents. Burrowing owl monitoring studies at Moffett Federal Airfield (Trulio 1994) and Mineta San José International Airport (Barclay 2007) show that the number of individuals observed declines from October to March. However, burrowing owls may not actually leave during this time (see banding summary below), but may just be less visible, as shown by LaFever et al. (2008) and suggested by Thomsen (1971) and Coulombe (1971) because they spend more daylight hours in their burrows. Trulio (1994) reported that the number of burrows used at Moffett Federal Airfield did not decline during the winter, suggesting owls are less visible during the winter months. In central California, burrowing owls occur only as winter

visitors in some coastal areas that appear to contain suitable breeding habitat (Garrett and Dunn 1981).

Recoveries of burrowing owls banded in California are another source of information about the nature of owl migration and dispersal. U.S. Geological Survey Bird Banding Laboratory records (through August 2003) contained 106 resightings of 4,708 burrowing owls banded in California (Barclay 2007). Seventy-five (71%) of these encounters occurred in the same 10-minute block of longitude and latitude (361 kilometers^2 or 139 miles^2) where the owls were banded, and 27 (25%) occurred in the 10-minute block adjacent to where they were banded. Of the remaining four encounters of burrowing owls that were banded and recovered in California, all were less than 95 kilometers from the block where they were banded (Barclay 2007).

Burrowing owls exhibit high site-fidelity and sometimes reuse burrows year after year, although dispersal distances may be considerable and variable depending on location and the age of the owls. Distances of approximately 53–150 kilometers (33–93 miles) have been observed in California for adult and natal dispersal, respectively (Gervais et al. 2008) but are usually much shorter (Table 3). Sizes of burrowing owl territories and home ranges also vary (Table 3). For example, at the Oakland Airport in California estimated breeding territories ranged from about 0.04 to 1.1 hectares (0.1–2.8 acres) (Thomsen 1971). Male ranges can be quite large, with estimated ranges as large as 3 kilometers^2 (740 acres) (Haug and Oliphant 1987).

Table 3. Spatial Information for Burrowing Owl

Type	Distance/Area	Location of Study	Citation
	May forage over 2–3 km^2 during nesting season	California agriculture; Saskatchewan agriculture	Rosenberg and Haley 2004; Haug and Oliphant 1987
Home range (male)	114 hectares (282 acres)	Imperial Valley, farm fields	Rosenberg and Haley 2004
	476 acres	San Joaquin Valley crop-grassland mosaic	Gervais et al. 2003

Table 3. Spatial Information for Burrowing Owl

Type	Distance/Area	Location of Study	Citation
	596 acres	Saskatchewan crop-grassland mosaic	Haug and Oliphant 1990
Breeding Territory	Range: 0.04–1.1 hectares (0.1–2.8 acres).	Oakland, California	Thomsen 1971
	Minimum: 7 acres	Desert in New Mexico	
Distance between Nest Burrows	Varies from 90 m to under 14 m	Idaho, Texas	Poulin et al. 2011, references therein
Dispersal	Juveniles disperse about 0.25 km (0.4 mi) from natal burrows after fledging.	Idaho	King and Belthoff 2001
	Adults disperse an average of 3.1 km (range 0.2–53 km)	Carrizo Plain, California	Rosier et al. 2006
Migration	Highly variable, little data; Most southern California birds are year-round residents	California and elsewhere	Poulin et al. 2011; DeSante et al. 1997; Harman and Barclay 1997

km – kilometer
m – meter

Ecological Relationships

In California, burrowing owls most commonly live in burrows created by ground squirrels (*Spermophilis* spp.) (Gervais et al. 2008). Therefore, the suitability and quality of burrowing owl habitat in the Plan Area is closely and positively related to the occurrence and population health of ground squirrels. Burrowing owls on the Great Plains depend mainly on prairie dogs for suitable burrows. In Great Basin sagebrush steppe, where ground squirrels do not occur, burrowing owls may depend on badgers (*Taxidea taxus*) for nest burrow excavation, although this species is a major predator of burrowing owls (Green and Anthony 1997). Burrowing owls prefer

grazed areas where livestock have reduced vegetation height (Wedgwood 1976). Green and Anthony (1989) found that nests lined with livestock dung were less prone to predation and had increased insect prey presence (Smith 2004), but uncertainty remains in the effect of grazing on burrowing owls and their habitat (Klute et al. 2003). In addition to badgers, native mammalian and avian predators include coyotes (*Canis latrans*) Swainson's hawks (*Buteo swainsoni*), ferruginous hawks (*B. regalis*), merlins (*Falco columbarius*), prairie falcons (*F. mexicanus*), peregrine falcons (*F. peregrinus*), great horned owls (*Bubo virginianus*), red-tailed hawks (*B. jamaicensis*), Cooper's hawks (*Accipiter cooperii*), and crows (*Corvus brachyrhynchos*) (Poulin et al. 2011). Non-native species, especially domestic dogs (*Canis familiaris*) and cats (*Felis domesticus*) are known predators of adult and young burrowing owls. Cannibalism has also been reported.

Population Status and Trends

Global: Stable (NatureServe 2010)
State: Declining; Priority 2 Species of Concern (Gervais et al. 2008)
Within Plan Area: Declining (Bloom 2009)

Recently published survey results based on a random sample of 860 5-kilometer^2 blocks in California in 2006–2007 yielded an estimate for the breeding-season population of burrowing owls of 9,187 pairs (±2,346 pairs) (Wilkerson and Siegel 2010). When comparing these results to 1993 results for the same survey areas using the same methods, the results indicate a population decline of approximately 10.9%, although the difference is not statistically significant. (The relatively large margin of error weakens the power of the test to show statistical differences.) Many regions in the Plan Area were not systematically surveyed prior to 2006–2007 (except for the Imperial Valley agricultural complex). Within the Plan Area agricultural development supports the highest densities of burrowing owls known in the world. However, a survey by Bloom Biological for the Imperial Irrigation District from 2007 to 2008 indicated a decline in the size of the Imperial Valley agricultural population (Bloom 2009). Population surveys are currently being conducted by the Imperial Irrigation District with results to be published in the summer of 2012 (Lovecchio,

pers. comm. 2012). This will help to determine if the decline recorded in 2007–2008 is in fact a longer-term trend.

There were no surveys for burrowing owls prior to 2007 in the West Mohave Desert. Once surveyed, the results yielded an estimate of 560 (±268) pairs of burrowing owls. Due to the survey's focus on a portion of the agricultural valleys, and the subsequent extrapolation of agricultural survey results to non-agricultural desert scrub areas of the West Mojave Desert, this number may constitute either a gross over-estimate or a gross under-estimate of the true number of burrowing owls in the region (Wilkerson and Siegel 2010). Just west of the Plan Area, 53 burrowing owls were found in the Coachella Valley during the 2006–2007 surveys. However, other areas in central-western Kern County (and Rosedale west of the Plan Area) were estimated to have lost at least 95 breeding pairs, since 1993, apparently related to expanding urban development on the west side of Bakersfield (Wilkerson and Siegel 2010).

Threats and Environmental Stressors

The most immediate threats to the burrowing owl are the conversion of grassland habitat to urban other than livestock grazing and the loss of agricultural hay, grass, and alfalfa lands to development or conversion to unsuitable crops like cotton, vineyards, orchards, corn and sugarcane (Gervais et al. 2008, Wilkerson and Siegel 2010). Vehicle collisions may also be a significant cause of mortality in the Plan Area (BLM 2005). All of these factors are well-established for burrowing owls in many parts of California (Gervais et al. 2008; Poulin et al. 2011; Hamilton and Willick 1996), and can be expected to increase in desert areas as a result of continuing regional human population growth and concomitant changes in land uses.

Associated with the habitat loss and degradation is the decline of fossorial species across much of the owl's historical range that create suitable nest sites for burrowing owls, such as ground squirrels, badgers, marmots (*Marmota* spp.), skunks (*Mephitis* spp., *Spilogale putorius*), kangaroo rats (*Dipodomys spectabilis*), and desert tortoises (*Gopherus agassizii*) (Gervais et al. 2008; Poulin et al. 2011). Eradication programs that have decimated rodent populations have, in turn, decreased the abundance of key prey available for burrowing owls. Because the

burrowing owl depends on other animals to dig its burrows, loss of fossorial species limits the extent of burrowing owl habitat across much of the Plan Area (Poulin et al. 2011).

Direct causes of mortality in burrowing owls include: predation by hawks, owls, badgers, coyotes foxes, domestic dogs and cats, and others (Poulin et al. 2011); vehicular collisions; wind turbines; barbed wire fences; shooting; road maintenance; tilling, pesticide application and other agricultural practices; and disease and parasites (Gervais et al. 2008; Poulin et al. 2011). Vehicular collisions, which accounted for 25 to 60% of burrowing owl mortalities in three studies (summarized in Poulin et al. 2011), are a significant cause of mortality because burrowing owls habitually perch and hunt on roadways at night (Bent 1938; Poulin et al. 2011). James and Fox (1987) were able to determine that reproductive success was directly proportional to the distance of pesticide application from burrows as a result of direct toxicity. Indirect mortality may also result from pesticide application to burrowing owl prey (James et al. 1990).

The fallowing of agricultural land in Imperial Valley as the water allocation to Imperial Valley Farms is reduced may produce less abundant habitat for rodents and invertebrates on which the burrowing owl preys. In some cases, losses to development are spurred on because of the loss of water for irrigating pastures. The robustness of the Imperial Valley burrowing owl population may be at risk if suitable agricultural habitat converts to habitat for renewable energy installations as landowners make economic decisions to shift land uses based on the potentially declining availability of irrigation water (Campbell, pers. comm. 2012).

Conservation and Management Activities

The burrowing owl is in decline across broad areas of its distribution in the United States and Canada. Several species status reviews, spanning a broad spatial scale from continental, to regional, to site- or project-specific have addressed the need for burrowing owl conservation and management. Broad-scale plan include *North American Conservation Action Plan, Western Burrowing Owl* (Commission for Environmental Cooperation 2005), *Status Assessment and Conservation Plan for the Western Burrowing Owl in*

the United States (Klute et al. 2003), *Recovery Strategy for the Burrowing Owl (*Athene cunicularia*) in Canada* (Environment Canada 2007), *Recovery Plan for the Burrowing Owl in Canada* (Hjertaas 1997), "Effects of Management Practices on Grassland Birds: Burrowing Owl" (Dechant et al. 2002), *Sonoran Joint Venture: Bird Conservation Plan, Version 1.0* (Sonoran Joint Venture Technical Committee 2006), and *The Desert Bird Conservation Plan: A Strategy for Protecting and Managing Desert Habitats and Associated Birds in California* (Bates 2006). The State of California has issued guidance on how development projects should mitigate impacts to burrowing owls (CDFG 2012). Recently issued conservation plans within the DRECP Area and adjacent desert regions are detailed in the West Mojave Plan (BLM 2005), the Imperial Irrigation District's 2009 Annual Water Report (Imperial Irrigation District 2010), and the CVMSHCP (CVAG et al. 2007). Habitat conservation planning efforts outside the DRECP Plan Area have also addressed the burrowing owl for example: East Contra Costa County HCP/NCCP (2006), Santa Clara Valley HCP/NCCP Draft (2012), and San Joaquin County Multi-Species Habitat Conservation and Open Space Plan (2000).

In California, the Department of Fish and Game, has completed the *Staff Report on Burrowing Owl Mitigation* (2012). This document provides guiding principles for conservation, conservation goals, and mitigation methods. The report includes habitat assessment and reporting details, breeding and non-breeding season survey and reports, a mitigation management plan and vegetation management goals.

Efforts to manage burrowing owls have employed a variety of techniques to address site-specific goals and conditions. Common management activities have addressed habitat management on preserve lands (Johnson 1986; Stanton and Teresa 2007; CVAG et al. 2007); evaluation of impacts from development projects (Bendix 2007; Smith and Belthoff 2001; Trulio 2001); prevention of disturbance during the nesting season (Koshear et al. 2007; (CVAG et al. 2007); installation of artificial burrows (Collins and Landry 1977; Poulin 2000; Smith and Conway 2005; Smith et al. 2005; Wildlife Research Institute, Inc 2005; Barclay 2008); and management of burrowing owls on military installations and airfields (Barclay 2007; Garcia and Conway 2007; Rosenberg et al. 1998, 2009; Trulio 2001). Other management efforts listed by Poulin et al. (2011) include

"installation of perches which provide hunting and predator observation sites; captive breeding and release; relocation of owls under immediate threat; pesticide restrictions; traffic and other warning signs; land stewardship agreements; and vegetation management through fire or grazing." Poulin et al. (2011) cites the highly successful use of artificial burrows by Olenick (1990) in Idaho.

The reintroduction of burrowing owls into vacant ranges has been done with limited success in British Columbia (Munro et al. 1984; Leupin and Low 2001), Manitoba (De Smet 1997), Minnesota (Martell et al. 2001), southwest Oregon (Green pers. comm.) and on a token, experimental scale in California (Delevoryas 1997). Because this species shows strong site fidelity to nesting areas, introducing birds to new areas is a challenge.

Management practices have also been implemented to address the unwanted occurrence of burrowing owls in some settings. These include passive relocation (Trulio 1995; Bendix 2007) and active relocation (Feeney 1997; Bloom et al. 2003) to remove burrowing owls from development project sites where impacts to occupied burrows were unavoidable and avoidance of direct take was desirable (Smith and Belthoff 2001). Management has also been carried out to address predation of burrowing owls on other special-status species (Garcia and Conway 2007). According to Lincer and Bloom (2007), burrowing owls were removed from areas between Camp Pendleton and Tijuana Slough National Wildlife Refuge (PHB) at potential California least tern and western snowy plover breeding sites.

Data Characterization

Parts of the Plan Area were randomly sampled for burrowing owl populations recently for the first time, including portions the Mojave and Sonoran deserts (Wilkerson and Siegel 2010). While this survey provides an objective statewide population estimate, and includes previously unsurveyed areas, it contains systematic sources of error and other limitations (e.g., the range in the extrapolated population estimate of 560 ±268 pairs for the western Mojave Desert is quite large). Potential sources of error include observer detection shortcomings, a lack of a detection probability assessment, untested population estimate assumptions (e.g., assuming a breeding pair when

observing a single owl), and large tracts of lands that remain unsurveyed due to access restrictions (which limits the ability to conduct a truly random sample). Although these limitations preclude a precise absolute population size estimate, the methods in the Wilkerson and Siegel (2010) survey adhered to those established by DeSante et al. (2007); the method of comparison between the two studies and thus the population trend estimates can be made. The information gained from these surveys informs the background of conservation planning for the burrowing owl in the DRECP Area. However, the potential sources of error identified above and the relatively weak statistical power to precisely estimate population size from the survey methods highlight the need for further census and monitoring efforts. Due to the high demographic variability of burrowing owl populations (Wilkerson and Siegel 2010) and prior documentation of burrowing owl population model inaccuracies (Johnson 1997), uncertainties remain in assessing the effect of conservation activities.

An ongoing data-collection protocol is specified in the Imperial Irrigation District's 2010 Annual Report (Bloom 2009).

Management and Monitoring Considerations

Though populations may be stable in areas such as the Imperial Valley and the western Mojave Desert, populations elsewhere in California have declined in numbers since the 1991–1993 survey, especially where agricultural land has converted to urban development (Wilkerson and Siegel 2010). Management and monitoring can be difficult since the vast majority of burrowing owl habitat in California occurs on private agricultural lands (Wilkerson and Siegel 2010). The Imperial Valley agricultural areas in the Plan Area support the most dense burrowing owl populations known anywhere, making conservation of the species especially challenging (Gervais et al. 2008).

Wilkerson and Siegel (2010) identified several important considerations for successful burrowing owl management, such as recognizing the species association with ground squirrels and agricultural water control infrastructure, and identifying the

particular conservation needs for newly surveyed populations in the western Mojave Desert. Preserving traditional nesting sites, as burrowing owls often reuse nesting sites occupied in recent years, is also an important management consideration (Dechant et al. 2002).

Pesticide use to control pest species in agricultural and urban-interface areas has clear adverse effects on burrowing owls due to direct mortality, weight loss, loss of animals that provide burrows, and loss of prey base (Poulin et al. 2011). Alternative integrated pest management strategies may be possible, though research on California ground squirrels (*Spermophilus beecheyi*) indicates that trapping and relocating is not a useful management alternative for problem ground squirrel colonies in most instances (BLM 2005; Van Vuren et al. 1997). Dechant (2002) provides recommendations for pest control that minimize negative impacts to burrowing owls, including excluding pesticide use around burrowing owl nests, restricting prairie dog control measures such as poisoning and baiting, and restricting the timing of pest control activities to avoid burrowing owl nesting, or nest selection periods.

Threats to burrowing owls associated with public land uses differ from those on private lands and, therefore, require different management considerations. On BLM-administered land, the Western Mojave Plan (2005) identified as primary short-term conservation needs reducing burrowing owl mortality from both on- and off-highway vehicle (OHV) collisions and protecting the species from shooting and harassment. In the long-term the Western Mohave Plan calls for occupied and potentially occupied habitat protection and for maintaining populations of fossorial mammals. Suggested management considerations in occupied and potential burrowing owl habitat on BLM lands included prohibiting OHV use and imposing speed limits, prohibiting certain pest control measures, educating recreational users, and requiring surveys prior to land-use changes. Livestock grazing may enhance habitat suitability by reducing vegetation height, and nests lined with livestock dung may reduce predation as well as increasing insect prey activity (Green and Anthony 1989; Smith 2004), but the effects of livestock on burrowing owls are not well documented and grazing management objectives may conflict with other habitat management objectives (e.g., managing for ground squirrel populations). The potential benefit of

livestock grazing on burrowing owl habitat on public lands would need to be tested.

Much remains to be learned about the kinds of habitat alteration tolerated by burrowing owls, including noise impacts and the duration and daily timing of nearby human activities (BLM 2005). The close proximity of some burrowing owl populations to airports suggest that they are tolerant of noise and other activities, but these relationships are not well understood (e.g., are these individuals just making the best of a marginal situation). Sustained population monitoring is important to assess the success of burrowing owl management practices because population levels can be highly variable, little information exists on the lifetime reproductive success of the species, and population trends have been difficult to predict in California (Johnson 1997; Poulin et al. 2011). Wilkerson and Siegel (2010) encouraged the engagement of "citizen-scientists" in ongoing monitoring efforts to reduce cost, expand monitoring scope, and increase awareness, but future surveys should incorporate latest research (e.g., Conway et al. 2008) to increase survey accuracy and population estimation precision.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for burrowing owl, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 6,496,668 acres of modeled suitable habitat for burrowing owl in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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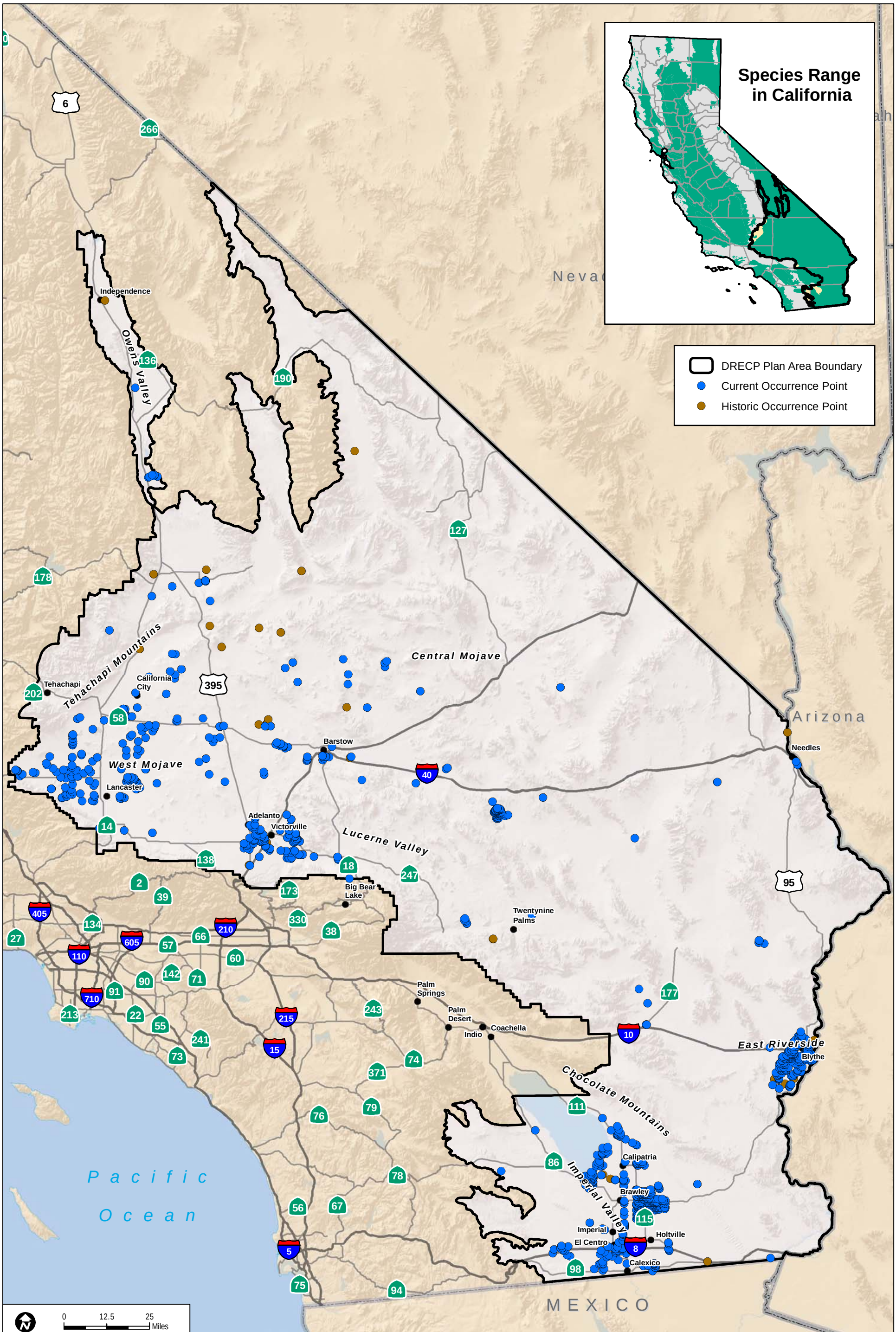

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B03

Burrowing Owl Occurrences in the Plan Area

California Black Rail (*Laterallus jamaicensis coturniculus*)

Courtesy of Peter La Tourrette © California Academy of Sciences.

Legal Status

State: Fully Protected; Threatened
Federal: Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A

Notes: A recent molecular genetic analysis (Girard et al. 2010) indicates that birds within and south of the Plan Area may qualify as a separate Distinct Population Segment (DPS) under the Endangered Species Act (ESA) (see Taxonomy section). No listing petition has ever been filed for this species (USFWS 2011), but this new information may result in reappraisal of the status of the species in the Lower Colorado River/Salton Trough region.

Taxonomy

The black rail (*Laterallus jamaicensis*) includes several subspecies which are largely disjunct in distribution. The two North American subspecies (the nominate *L. j. jamaicensis* and the California black rail [*L. j. coturniculus*]) are widely accepted, while two of the three South American subspecies, Junin rail (*L. j.* or *L. tuerosi*) and Galapagos rail (*L. j.* or *L. spilonotus*), are often regarded as separate species. Recent molecular analysis has revealed strong genetic divergence between coastal California, Central Valley, and Lower Colorado/Salton Trough populations (Girard et al. 2010). There is evidence for substantial gene flow between the coastal and Central Valley groups, but the Lower Colorado/Salton Trough group, "has a unique and highly divergent genetic composition" and may not have originated from the Coastal/Central Valley populations (Girard et al. 2010). Thus, it may constitute a separate subspecies and/or a "Distinct Population Segment" for the purposes of assessment and potential protection under the federal ESA.

Distribution

General

The California black rail occurs in California, Arizona, Baja California, and the Colorado River delta in Sonora. Figure SP-B05 shows the distribution of California black rail in the Plan Area. The subspecies appears to be composed of three clearly distinct populations. The coastal population is most numerous and inhabits tidal marshes mainly in the northern San Francisco Bay area, with smaller occurrences at sites from Bodega Bay to northwest Baja California. The intermediate-sized Central Valley population occurs at interior wetlands of Butte, Nevada, Placer, San Joaquin, and Yuba counties. The much smaller Lower Colorado/Salton Trough population primarily occurs at the following locations: (1) from Laguna Dam to Martinez Lake, Arizona; (2) around the Bill Williams River delta; (3) in the Colorado River delta area; and (4) in the Imperial Valley and adjacent Salton Sea (Eddleman et al. 1994; Patten et al. 2003, Hinojosa-Huerta, et al. 2004, Conway and Sulzman 2007, and Girard et al. 2010).

Distribution and Occurrences within the Plan Area

Historical

Grinnell and Miller (1944, pp. 130–131) were not aware of any occurrence of black rails in the Lower Colorado River/Salton Trough area, and the first report from the region was for an occurrence at Calipatria in the Imperial Valley (Laughlin 1947). It is thus possible that the rail was rare or absent from the Plan Area prior to construction of Colorado River dams, water diversions, and formation of the Salton Sea in 1905 (Patten et al. 2003). Extensive breeding season surveys were conducted in the area by Evens et al. (1991), at 906 stations in the Lower Colorado River and Salton Trough. They had 116 detections, with 65% of detections on the Lower Colorado River, 15% in seeps along the All American Canal, 12% at the Salton Sea, 7% at seeps along the Coachella Canal, and 1% at Finney Lake in the Imperial Valley. Overall, there are approximately 11 historical (i.e., pre-1990) California black rail occurrence records in the Plan Area (CDFW 2013; Dudek 2013). These occurrences are located in Imperial County, east of the Salton Sea (Figure SP-B05).

Recent

Extensive surveys in the southwestern U.S. in 2000 and 2001 largely confirmed the distribution found earlier, but found far fewer birds despite a greater survey effort, with populations at all sites stable or declining; most individuals were also in Arizona (Conway and Sulzman 2007). Currently, there are approximately 39 recent (i.e., since 1990) California black rail occurrences in the Plan Area. Recent occurrences of black rail in the Plan Area are primarily along the Lower Colorado River from the Laguna Diversion Dam upstream to about the head of Ferguson Lake (CDFW 2013; Figure SP-B05), although two more isolated occurrences extend the species' range along the river upstream to near Parker.

Other occurrences in the southeastern portion of the Plan Area include an isolated riparian marsh on the north side of the Salton Sea at the Dos Palmas Preserve Area of Critical Environmental Concern on Bureau of Land Management (BLM) lands, which is supported by seepage from the Coachella Canal; a marsh on the New River near Seeley; marshes at the mouth of the river where it enters the Salton Sea; and marshes supported by seepage from the All American Canal southeast of El Centro (Conway and Sulzman 2007).

In the northern portion of the Plan Area the species has been recorded at Little Lake (Inyo County 1964). In the southwestern portion of the Plan Area, the species was discovered as a suspected breeder at a Carrizo Marsh in Anza Borrego Desert State Park (San Diego County) in 1974 and 1976, but the marsh habitat was destroyed in September 1976 by tropical storm Kathleen and replaced by tamarisk (*Tamarix* spp.); there are no subsequent records for black rail in this area since 1976 (Unitt 2004). Single detections at Big Morongo Preserve in May 1983 and November 1984 suggest an attempt to establish there; the potential is substantial for small, undetected populations at other locations in the Plan Area (Campbell, pers. comm. 2012).

Natural History

Habitat Requirements

Suitable California black rail habitat generally includes salt marshes, freshwater marshes, and wet meadows. Most or all southwestern U.S. populations are nonmigratory, and these habitat types serve for breeding, foraging, and overwintering.

During the most recent comprehensive survey of California black rail occurrence in the southwestern U.S., Conway and Sulzman (2007) found all sites with black rail detections in riparian marsh habitat. At many sites, upland habitat (chiefly Mojave or Sonoran desert lowland vegetation) or open water were present within 50 meters (164 feet) of the detection site. Vegetation was compared between sites with and without black rails. Species positively correlated with black rails were common threesquare (*Schoenoplectus pungens*), arrowweed (*Pluchea sericea*), Fremont cottonwood (*Populus fremontii*), and seepwillow (*Baccharis salicifolia*). These plants, in turn, are strongly associated with shallow water or moist soil near the upland/wetland interface. Similar results were reported from prior surveys in the region, with Evens et al. (1991) reporting the species most frequent at occupied sites as common threesquare, cattails (*Typha angustifolia* and *T. domingensis*), California bulrush (*Scirpus californicus*), and native tree/shrub communities. Tamarisk presence was also positively associated with black rails but the species was infrequent where tamarisk cover was 67% or greater (Conway and Sulzman 2007). Conway and Sulzman (2007) concurred with previous authors in further concluding that black rail was positively associated with sites that have very shallow standing water (less than 3 centimeters (1.18 inches) deep) and very low daily water level fluctuations.

Foraging Requirements

California black rails forage in the same habitats they use for breeding. They prey on small (<1 centimeter [0.39 inch]) invertebrates, chiefly insects, gleaned from marsh vegetation and mudflats; they also eat small seeds (Eddleman et al. 1994). Analysis of seven incidentally taken rails from an Arizona site found that the birds' diet included various beetles, grasshoppers, ants, earwigs,

spiders, and other miscellaneous arthropods, as well as snails, bulrush, and cattail seeds. Bulrush and cattail seeds appear to be an important component of their diet during the winter months when insect prey availability is low (Flores and Eddleman 1991, as cited in Eddleman et al. 1994).

Reproduction

The black rail reproductive cycle begins with pair formation (Table 1). Associated behavior has not been observed but may involve calls by both sexes, which have been recorded from late February into July on sites along the Lower Colorado River (Eddleman et al. 1994). Multiple broods may be raised; nest records from Arizona indicate that the peak of egg-laying for the first brood of the season is May 1 (Eddleman et al. 1994). One study of black rail nesting along the Lower Colorado River determined that located nests had a mean clutch size of 4.8 eggs (Flores and Eddleman 1993). Nests were in clumps of vegetation elevated an average of 6.4 centimeters (2.52 inches) above the mud substrate. Incubation began at varying dates from March 30 to June 25, lasting from 17 to 20 days. Both sexes incubated the eggs. The birds aggressively defended the nests by scolding, raising their wings, and running toward researchers. Both young and parents abandoned the nest within 24 hours after the last egg in each clutch had hatched. Newborn hatchlings, although fairly precocious, are small and downy; it appears likely a period of parental care is needed, but there are no data on the subject (Eddleman et al. 1994). One female was recaptured 18 days after nest abandonment with an egg in her oviduct, suggesting that multiple brooding may occur (Flores and Eddleman 1993).

Table 1. Key Seasonal Periods for California Black Rail

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding		X	X	X	X	X	X					

Source: Eddleman et al. 1994

Repking and Ohmart (1977) reported California black rail densities of 1.14 to 1.58 calling birds per hectare (0.46 to 0.64 calling birds per acre) in spring, and 0.73 birds per hectare (0.29 birds per acre) in winter, on the lower Colorado River. In Arizona, black rails used home ranges averaging 0.4 ±0.2 hectare (0.98 ±0.49 acre) and rarely overlapped (Flores 1991, as cited in Harvey et al. 1999).

Spatial Behavior

Movement of rails is primarily by running along the ground, often using trails made by voles (*Microtus* spp.). Rails can also swim short distances. Flight, which exposes them to aerial predators, is uncommon (Eddleman et al. 1994).

California black rails are believed to be nonmigratory, but their occurrence at many small locations indicates that dispersal movements occur (Eddleman et al. 1994). However, there is no documentation of the timing or manner of such movements.

Ecological Relationships

Black rail predators have not yet been identified in the Lower Colorado River/Salton Trough region. Elsewhere, documented avian predators include great blue heron (*Ardea herodias*), great egret (*Casmerodius albus*), northern harrier (*Circus cyaneus*), ring-billed gull (*Larus delawarensis*), great horned owl (*Bubo virginianus*), and short-eared owl (*Asio flammeus*) (Eddleman et al. 1994). Known mammalian predators include rats (*Rattus* spp.), red fox (*Vulpes vulpes*), and domestic cats (*Felis domesticus*). Nest predators likely include a variety of other mammals and reptiles as well (Eddleman et al. 1994).

Little is known about competition among black rails or between black rails and other species. Richmond et al. (2010), investigating competition between California black and Virginia rails in Northern California freshwater marshes, found a positive association between the two species; in the smallest marshes, Virginia rail presence was a good predictor of black rail presence.

Brood parasitism of black rails is not known to occur (Eddleman et al. 1994). It is likely that black rails, as most birds, are subject to infectious

disease and to parasitism by invertebrates such as mites and protozoans, but this has not been documented (Eddleman et al. 1994).

Mutualistic or commensal relationships do not appear to have been identified in black rails.

Population Status and Trends

Global: Declining (Birdlife International 2008)
State: Declining (Conway and Sulzman 2007)
Within Plan Area: No formal assessment, but results of Evens et al. (1991) and Conway and Sulzman (2007) strongly indicate populations are declining.

Comprehensive surveys of California black rail distribution and status were performed for the Lower Colorado River/Salton Trough region in 1973–1974 (Repking and Ohmart 1977), 1988-1989 (Evens et al. 1991), and in 2000–2001 (results included in Conway and Sulzman 2007). Repking and Ohmart (1977) found 106 birds in 1973 and 100 in 1974. Evens et al. (1991) found 75 birds in 1989. Conway and Sulzman (2007), in the most comprehensive survey effort of this region to date, report 136 birds in 2000-2001 surveys, including 100 along the Lower Colorado River, mostly in marshes between Laguna Dam north to Ferguson and Martinez Lakes, 21 black rails at three marshes along the All-American Canal. Of the 100 black rails detected along the Lower Colorado River, 38 were in the Plan Area in California (Conway et al. 2002, as cited in Corman and Wise-Gervaise 2005).

The 1991 study (Evens et al. 1991) reported that "subpopulations were small and isolated" and that "[t]he causes of this downward trend—all related to habitat loss or degradation—are pervasive and ongoing". Conway and Sulzman (2007, p. 996) delivered a similar conclusion: "Our data suggest that degradation and elimination of suitable emergent marshes over the past 25 to 30 years has caused significant reduction in black rail distribution in Southern California and Arizona."

Threats and Environmental Stressors

Human impacts on black rails include shooting and trapping, contaminants, collisions, effects of research, and habitat impairment. Shooting and trapping effects in modern times are likely very minor due to the small size of the bird (Eddleman et al. 1994). Contaminant effects, such as from exposure to pesticides, are virtually unknown, but slightly elevated selenium levels were found in Lower Colorado River birds and eggs analyzed in 1988 (Flores and Eddleman 1991, as cited in Eddleman et al. 1994). The habitat requirement for shallow wetlands makes California black rails especially vulnerable to manipulations of water levels in what are now heavily managed to entirely human-created environments. Research effects include potential disturbance of nesting birds during surveys, and more severe effects, such as mortality, nest failure, or exposure to predation, may occur in association with mist netting, radio tracking, or other invasive research techniques.

Specifically addressing the Lower Colorado River/Salton Trough populations, Conway and Sulzman (2007) identify degradation and loss of suitable emergent marsh habitat as the principal threat to the species. They also note declines in habitat suitability due to the spread of tamarisk.

Conservation and Management Activities

California black rail is not the subject of a documented recovery plan, and there do not appear to be any active state or local programs focused on its conservation and management. However, it is a covered species in several approved Habitat Conservation Plans (HCPs) and Natural Community Conservation Plans (NCCPs). Several of these only affect the coastal and/or Central Valley populations and are not related to the Desert Renewable Energy Conservation Plan (DRECP) area. However, the rail is a covered species under both the Coachella Valley Multiple Species Habitat Conservation Plan (CVMSHCP) and the Lower Colorado River Multi-Species Conservation Program (LCRMSCP). Both the CVMSHCP and LCRMSCP include provisions to create or enhance black rail habitat within the proposed DRECP area. The CVMSHCP includes conservation and creation of black rail habitat at several sites in its plan area, as well as broader conservation

actions such as control of tamarisk and measures to ensure proper hydrologic function of conserved habitat (CVAG 2007, pp. 9-132 to 9-137). The LCPMSCP includes provisions to maintain existing black rail habitat and to create new habitat along the Lower Colorado River (LCRMSCP 2004, pp. 5-57 to 5-58).

Data Characterization

Although the black rail is very difficult to detect, its general habitat requirements are well understood, and it remains within a small home range in suitable habitat. Most currently occupied areas have benefitted from past alterations or creation, albeit without human intent to create habitat for the species. Accordingly, it is feasible to identify, conserve, or even create habitat that will be used by black rails. A key obstacle to black rail management is a complete absence of quantitative knowledge regarding dispersal movements. Past surveys (Evens et al. 1991; Conway and Sulzman 2007) have documented disappearance of black rails from apparently suitable habitat without recolonization. Population models applied to black rail sites in the Central Valley predict that the existing small, dispersed populations (or demes) are not large enough to be self-sustaining (Girard et al. 2010). However, these small populations have persisted, suggesting that birds are moving to and/or among these populations in a manner that is not yet understood. Resolving such population dynamics is a prerequisite to successful black rail recovery effort in California populations.

Management and Monitoring Considerations

Black rail management at existing preserves along the Lower Colorado River, such as the Bill Williams River National Wildlife Refuge and the Mittry Lake Wildlife Area (both in Arizona), as well as under approved HCPs such as the Coachella Valley MSCP and the Lower Colorado River MSCP, focuses on conserving and maintaining suitable habitat conditions by maintaining suitable hydrology and plant communities.

Any management actions potentially affecting California black rail habitat would likely require surveys to assess the potential for habitat occupancy. Survey protocols appropriate for habitat in the Lower

Colorado River/Salton Trough area have been developed and are described by Conway (2005) with additional information available at the North American Marsh Bird Monitoring Program website (http://www.cals.arizona.edu/research/azfwru/NationalMarshBird); this protocol is currently used for the Lower Colorado River MSCP.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for California black rail, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 220,888 acres of modeled suitable habitat for California black rail in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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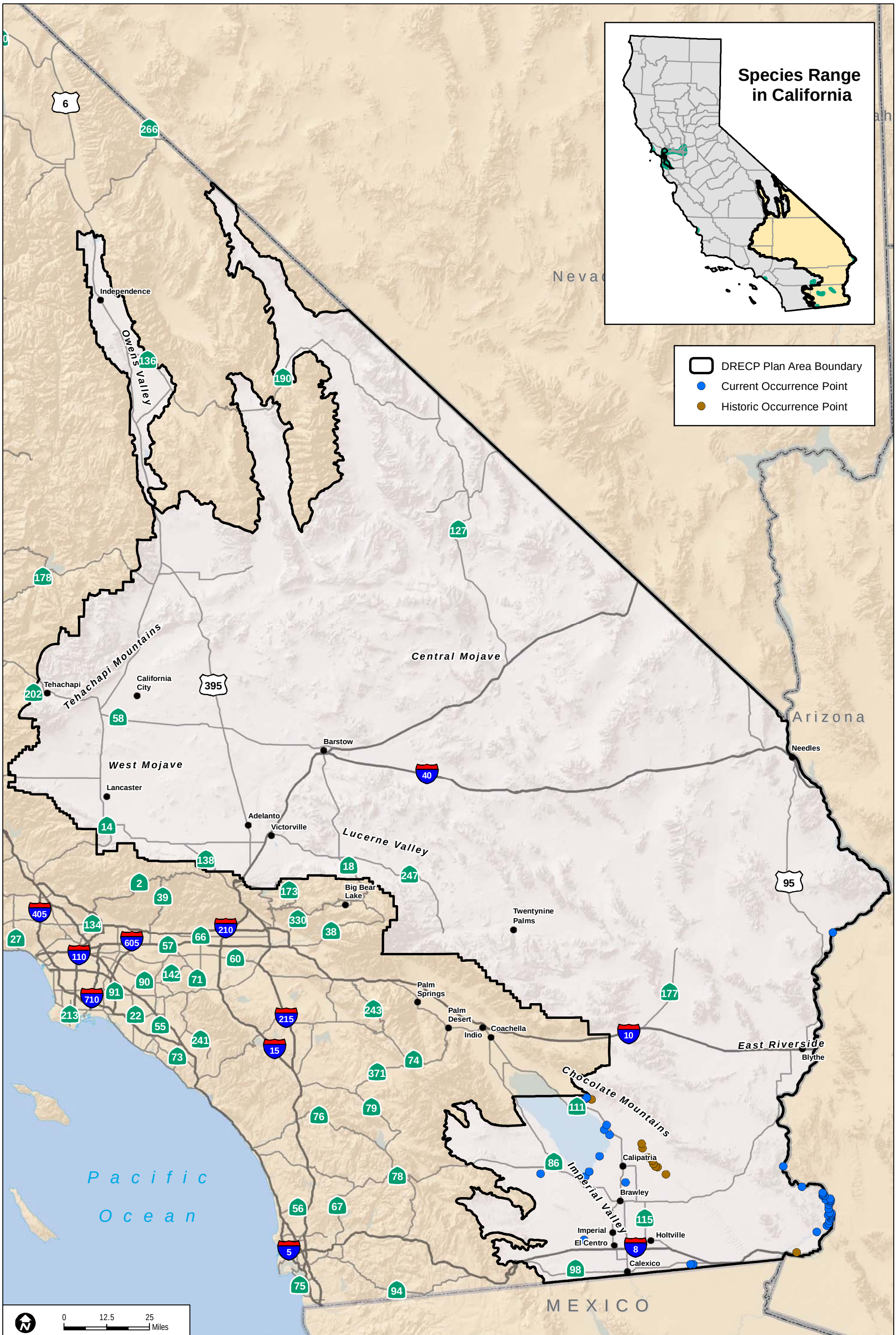

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B04

California Black Rail Occurrences in the Plan Area

California Condor (*Gymnogyps californianus*)

Photo by Dudek.

Legal Status

State: Endangered, Fully Protected
Federal: Endangered
Critical Habitat: Critical habitat was originally designated on September 24, 1976 (41 FR 41914–41916) and revised the following year on September 22, 1977 (42 FR 47840–47845).
Recovery Planning: The latest version of the recovery plan for this species has been completed (U.S. Fish and Wildlife Service 1996).
Notes: Spotlight Species Action Plan 2010–2014 has been completed (U.S. Fish and Wildlife Service 2009). The USFWS 5-year Review was completed in June 2013 (USFWS 2013a).

Taxonomy

The California condor is a member of the family Cathartidae, or New World vultures that consist of seven species ranging throughout most of North and South America (Houston 1994). Although similar to the 15 species of Old World vultures that occur in Africa, Europe, and Asia, Old World vultures belong to the family Accipitridae, which includes eagles, hawks, kites, and buzzards. These groups have evolved from different lineages and are a well-known example of convergent evolution (Sibley and Ahlquist 1990; Houston 1994). The California condor is a close relative of the Andean condor (*Vultur gryphus*) that inhabits western coastal and mountainous portions of South America.

Distribution

General

Knowledge of the prehistoric and historical range of the California condor comes from fossil records, Native American feather regalia, and written records. Archaeological evidence suggests that during the Pleistocene era condors existed on both coasts of North America, but primarily occupied the west coast (Snyder and Snyder 2000; D'Elia and

Haig 2013). Fossil evidence from New Mexico, Arizona, Utah, a single site in New York, sections of northern Mexico, and southern Canada support this hypothesis (Hansel-Kuehn 2003; Brasso and Emslie 2006). By 1800, California condors were restricted to their west coast range, which stretched from British Columbia, Canada, to Baja California, Mexico, with small inland populations in regions such as the Grand Canyon (Snyder and Snyder 2000; D'Elia and Haig 2013). Condors were in the Pacific Northwest until the beginning of the twentieth century and found in the southern segment (Baja California) until the 1930s (Koford 1953; Wilbur 1973). By the middle of the twentieth century, condors were confined to a small region in Southern California. (Figure SP-B06). From the late 1970s to 1987 when the last few condors were trapped for captive breeding purposes, condors foraged primarily in the foothills bordering the southern San Joaquin Valley and valleys in San Luis Obispo, Santa Barbara, Kern, and Tulare counties.

Currently, the condor is found in three disjunct populations: a reintroduced population in both Southern and central–coastal California, a reintroduced population in the Grand Canyon area of Arizona, and a reintroduced population in Baja, California, Mexico.

Distribution and Occurrences within the Plan Area

Historical

In California by the middle of the twentieth century, condors had declined to the extent that they only occurred in a wishbone-shaped area encompassing 10 counties north of Los Angeles, California, including San Benito, Monterey, San Luis Obispo, Santa Barbara, Kern, Ventura, Tulare, Fresno, Kings, and Los Angeles counties (Wilbur 1978). Historical sightings in the Plan Area were primarily in the northwestern portion of the Plan Area in the area around Tehachapi. Some historical sightings were east of the Piute Mountains, south and east of Bright Star and along the western edge of Red Rock Canyon.
Farther south, there is a historical occurrence along the southwestern boundary of the Plan Area northeast of Acton and one southwest of Lancaster (Figure SP-B06).

Recent

By 1987, the last individuals were trapped out of the wild for captive breeding. Since 1992, releases of captive-bred individuals have

occurred in parts of California; Arizona; and Baja California, Mexico (San Pedro Martir Mountains). The California condor occurs principally along the western edges of the Desert Renewable Energy Conservation Plan (DRECP) area, specifically within the Tehachapi Mountains east of Interstate 5 and portions of the Los Padres National Forest west of Interstate 5 (USFWS 2010). Global Positioning System (GPS) data from the USFWS for 2003–2013 show 818 records for the Plan Area (Figure SP-B06). Most records are in and around Tehachapi. There are also records north of Hwy 14 and west of Red Rock Canyon. Along the southwestern boundary of the Plan Area there are records from the Northern Transverse Ranges, west and south of Quartz Hill, and east of Soledad Canyon (Figure SP-B06). It should be noted that as a rapidly expanding cumulative database, additional GPS records for the western edge of the Plan Area are expected. At this time, nesting has not been documented in the DRECP Plan Area; condor use of the Plan Area is currently limited to foraging and temporary roosting.

Figure 1 Range of the California Condor in the United States

Natural History

Nest Habitat Requirements

California condors were historically found in habitat with requisite populations of ungulates and other large vertebrates (Koford 1953; Snyder and Snyder 2000; Grantham 2007a).

California condors are primarily a cavity nesting species and typically nest in cavities located on steep rock formations or in the burned out hollows of old-growth conifers (coast redwood (*Sequoia sempervirens*) and giant sequoia trees (*Sequoiadendron giganteum*)) (Koford 1953; Snyder et al. 1986). Less typical nest sites include cliff ledges, cupped broken tops of old-growth conifers, and in several instances, nests of other species (Snyder et al. 1986; USFWS 1996). Key characteristics of a suitable nest site are that it is in a location at least partially sheltered from the weather and in a location easily approachable from the air, such as on a cliff, steep slope, or tall tree (Snyder et al 1986).

Foraging Habitat Requirements

California condors are obligate scavengers, feeding only on the carcasses of dead animals, primarily medium- to large-sized mammals, but also occasionally on reptiles and birds (Koford 1953, Wilbur 1978). Condor food items within interior California in prehistoric times probably included mule deer (*Odocoileus hemionus*), tule elk (*Cervus elaphus nannodes*), pronghorn antelope (*Antilocapra americana*), and smaller mammals. Along the Pacific shore, the diet also included whales, sea lions, and other marine species (Harris 1941; Koford 1953; Emslie 1987; FWS 1996). Koford (1953) estimated that 95% of the California condor diet consisted of cattle, domestic sheep, ground squirrels (*Spermophilus beecheyi*), mule deer, and horses. Recently, condors have been found to feed primarily on domestic animals (e.g., cattle), hunter-killed mule deer (*Odocoileus hemionus*) and wild pigs, shot or poisoned coyotes (*Canis latrans*), and ground squirrels (*Spermophilus* spp.).

Condors locate carcasses by eyesight, not olfaction, and may rely on watching other scavengers, especially turkey vultures (*Cathartes*

aura), golden eagles (*Aquila chrysaetos*), and common ravens (*Corvus corax*), to locate much of their food.

Most California condor foraging occurs in open terrain of foothill grassland and oak savanna habitats, and occasionally open scrub habitat. In the central coastal portion of the state, coastal plains and beaches are also suitable foraging habitat.

As large scavengers, California condors are evolutionarily adapted for feeding on the carcasses of deer, elk, whales, mastodons, and other large animals more prevalent in the Pleistocene (Emslie 1988). As such, the availability of large dead prey was often unpredictable, leading condors to develop a wide-ranging search behavior. Foraging flights occurred, and continue to occur, over vast areas encompassing hundreds of linear miles of travel each day (Meretsky and Snyder 1992). Condors tend to forage within 50 to 70 kilometers (km) (31 to 44 miles) of nests, but may travel up to 180 km (112 miles) in search of food. Core foraging areas for nesting birds range from about 2,500 to 2,800 km^2 (965 to 1,081 $miles^2$) (Meretsky and Snyder 1992). Non-breeding birds may have foraging ranges of 5,000 km^2 (1,930 $miles^2$) (USFWS 1996).

Like most scavenging birds, California condors are opportunistic. As such, individual birds may be expected to take advantage of local abundance of food almost anywhere within their normal range. Foraging behavior shifts may result from seasonal changes in climatic conditions (e.g., fog, thermal activity, wind intensities, rain) and from changes in food availability (Wilbur 1978).

Reproduction

Condors reach sexual maturity at the age of 5 to 8 years, and a captive male has successfully bred at age 5 (USFWS 1996). Pairs form in late fall and early winter, and remain together year-round and for multiple years. Nest prospecting generally occurs in January or February, several weeks before egg laying (Snyder and Schmitt 2002).

Clutch size is one egg, and a second clutch may be laid if the first fails early in the nesting season. First eggs are laid between the last week of January and the first week of April. The incubation period lasts an

average of 57 days, ranging from 53 to 60 days. Both sexes incubate, with shifts lasting several days in length. Chicks hatch from the last week of March through the first week of June. Chick brooding is nearly constant for the first 2 weeks after hatching, after which it declines and ceases during the day at about 1 month of age. Chicks are known to leave the nest cavity and scramble around on foot before taking their first flight. Fledging flights take place when chicks are 5.5 to 6 months old (early September to mid-November). Young are fully dependent on adults for about 6 months after fledging, and partial dependency continues for another 6 months (Snyder and Schmitt 2002). It was formerly thought that pairs nested only every other year because of the long period of parental care, but this pattern seems to relate to timing of successful fledging the previous year; if a nestling fledges early in the year (e.g., late summer–early fall), the pair may attempt nesting the following year (USFWS 1996).

Table 1. Key Seasonal Periods for California Condor Reproduction

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Nest prospecting	✓	✓										
Eggs	✓	✓	✓	✓	✓							
Nestlings			✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Dependent fledglings	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓

Notes: Active year-round resident

Source: Snyder and Schmitt 2002

Spatial Behavior

Spatial behavior by condors includes distances between nest sites, daily movements, and temporary movements for foraging and habitat-use patterns (e.g., individual foraging ranges) (see Table 2).

California condors are not migratory, though they are known to travel long distances during foraging flights as described above. One California condor traveled 141 miles (mi) 225 kilometers (km) in a

single day, from the northeast corner of Tulare County south through the Sierra Nevada mountain range and Tehachapi Mountains to a roost just north of the Santa Barbara nesting area (Snyder and Snyder 2000). Telemetry data and GPS devices on some birds have documented other long-distance flights, including flights from southern Utah to Flaming Gorge, Wyoming (over 400 mi (643 km) and from Sierra de San Pedro Martir in Baja California to Imperial County, California (approximately 155 mi (250 km) (USFWS, unpubl. GPS telemetry data). Studies conducted during the 1980s, as summarized by Meretsky and Snyder (1992), showed that the last California condors remaining in the wild prior to 1987 comprised a single population of birds occupying an area of approximately 2 million ha. (4,942,000 ac.). Insofar as could be determined, every California condor in the wild used the entire area and was capable of soaring between any two points within the area in a single day.

California condors use topography and associated thermal weather patterns for flight. In Southern California, both short- and long-distance flights have been shown to follow routes over the foothills and mountains bordering the southern San Joaquin Valley, avoiding passing directly over the flat valley. As an example, a condor heading to Tulare County from the coastal mountains of Santa Barbara County would cross northern Ventura County, travel through the Tehachapi Mountains in southern Kern County, then turn north to pass by Breckenridge Mountain, and enter Tulare County between the Greenhorn Mountains and Blue Mountain. Condors have also been observed flying over areas with less extensive flat agricultural regions (Cuyama Valley in Santa Barbara and San Luis Obispo Counties) (USFWS 1996).

Condors are dependent on uplift created by thermal cells or topographic relief features for soaring flight. Consequently, most foraging flights tend to occur in mountainous areas where winds deflected by hills provide uplift (Snyder and Schmitt 2002).

Extended flight is achieved by soaring, either gliding in uplifts along topographic features or circling for altitude in thermals, then losing altitude in long glides. Typical flight speed averages about 31 miles per hour (mph), but can reach 43 mph in long extended flights, depending on wind conditions. Condors' high wing-loading (weight-

to-wing area ratio; 7.7 kilograms/meters^2), which reduces condors' maneuverability, may explain their reluctance to forage over the flat bottom of the San Joaquin Valley and their tendency to forage later in the morning and earlier in the evening (when they will have optimum visibility) (Snyder and Schmitt 2002). This may also have prevented them from occupying the Midwestern U.S. and large portions of the Intermountain Region.

A recent analysis of global positioning system (GPS) data for the period of 2004 through 2009 shows that condor ranges in the Southern California population are becoming increasingly multimodal, with 2009 use concentrated in the Hopper Mountain and Bitter Creek NWRs, Wind Wolves Preserve, and Tejon Ranch, the latter of which exhibits recolonization for foraging purposes (Johnson et al. 2010). These recent GPS movement data indicate that condors are re-establishing foraging ranges that are consistent with their ranges prior to extirpation/removal from the wild in 1987 (Johnson et al. 2010).

Table 2. Movement Distances for California Condor

Type	Distance/Area	Location of Study	Citation
Distance between active nest sites	Nest sites as close as 0.5 miles apart	California	USFWS 1996
Territory	Not territorial except at nest	Southern California	Snyder and Schmitt 2002
Foraging range, breeding	31–44 miles from nest	Southern California	Meretsky and Snyder 1992
Foraging range, non-breeding	Up to 141 miles in a day or 700,000 hectares	Southern California	Meretsky and Snyder 1992

Ecological Relationships

California condors are principally scavengers. They range over vast areas in search of carcasses to feed on. As such, they are in competition with other scavengers and opportunistic carnivores. Such species might include other birds of prey (e.g., eagles, hawks), turkey vultures, the common raven, and American crow (*Corvus brachyrhynchos*), as well as

mammalian scavengers such as coyotes (*Canis latrans*), American badgers (*Taxidea taxis*), and weasels and skunks.

Since condors reside at the top of the food web (tertiary consumers), adult condors are mostly free from predation. However, nests and eggs are subject to predation by other birds of prey. Should nests be insufficiently isolated, they may also be subject to predation by bears, coyotes, foxes, and other mammalian predators.

Population Status and Trends

Studies from the 1930s to 1950 gave a population estimate of 60 to 100 condors (Robinson 1939, 1940; Koford 1953), though other evidence and further analysis suggests a more likely population size in 1950 of 150 individuals (Snyder and Johnson 1985). Using Koford's estimate of population size (1953), Miller et al. (1965) estimated only 42 birds were left in the wild in the early 1960s. In 1978, the wild population was estimated at 30 individuals (Wilbur 1980). Comprehensive counts of California condors began in 1982, with the advent of photo-censusing efforts allowing reliable identification of individuals (Snyder and Johnson 1985). This effort confirmed that the wild population declined from an estimate of 21 individuals in 1982, to 19 individuals in 1983, 15 individuals in 1984, and 9 individuals in 1985. The decline in the wild during this period resulted partly from the removal of birds for captive breeding purposes. By the end of 1986, all but two wild California condors had been taken into captivity. On April 19, 1987, the last wild California condor was captured and taken to the San Diego Wild Animal Park. At that time, there were 27 individuals in the global population.

Beginning in 1992, captive condors began to be released back into the wild, with increasing numbers being released in succeeding years. As of August 31, 2013, there were 424 California condors in the world population, including 201 in captivity and 223 in the wild (USFWS 2013b). The wild population includes 123 in central and Southern California, of which approximately 56 (not including 6 young still in the nest) currently inhabit Southern California and have the potential to visit portions of the Plan Area. The remaining wild population includes 30 birds in Baja California and 70 in Arizona. Due to a

combination of captive breeding and release, and wild nest reproduction, this population is steadily increasing and is expected to continue to increase, barring stochastic catastrophes.

Table 3. Numbers of California Condors in the Wild in August 2013

Location	Type	Number
Southern California	Wild-fledged	10
	Released free-flying	56
Central California[1]	Wild-fledged	11
	Released free-flying	61
Arizona	Wild-fledged	7
	Released free-flying	66
Mexico	Wild-fledged	2
	Released free-flying	29
Total		**213**

[1] Central California includes Pinnacles National Monument and Central Coast.

Source: USFWS 2013b.

Threats and Environmental Stressors

Because California condors are characterized by high survival rates and low reproductive rates, low rates of adult mortality are important for population stability (Meretsky et al. 2000; Snyder and Schmitt 2002; Walters et al. 2008). Condors have a clutch size of one egg, a normal nest success rate of 40%–50%, and an age of first breeding from about 5 to 8 years (USFWS 1996). They may nest in successive years if nestlings successfully fledge early in the year, but they usually skip years (USFWS 1996).

The decline of the condor population during the early 1900s has not been definitively linked to any particular cause; however, it was likely the result of high mortality rates due to direct persecution, collection of specimens, and secondary poisoning from varmint control efforts and 1,1,1-trichloro-2,2-bis(pchloro-phenylethane (DDT) (Snyder and Snyder 2005; D'Elia and Haig 2013). Lead poisoning may have been a contributing factor, but was not recognized as such until after 1980, at

which time it became identified as a major cause of mortality that resulted in the recent decline (Janssen et al. 1986; Bloom et al. 1989; Pattee et al. 1990; Cade 2007; Grantham 2007b; Hall et al. 2007), particularly since the development of lead ammunition that fragments upon impact in living tissue. In both California and Arizona, many reintroduced birds have been exposed to high levels of lead (Fry, 2003 and 2004; Cade 2007; Grantham 2007b; Hall et al. 2007; Hunt et al. 2007; Sullivan et al. 2007; Woods et al. 2007). Other recent documented sources of mortality include predation, powerline collision, micro-trash, fire, and shooting (USFWS 2013a).

The latest version of the Condor Recovery Plan (FWS 1996) suggests that habitat loss is not an important factor in the recovery of the condor. Similarly, Snyder (2007) did not identify habitat loss as a limiting factor for wild California condors. Although historical condor habitat, especially foraging areas, has been modified, condors are opportunistic scavengers and have switched from natural carrion to feeding on domestic livestock carrion with the conversion of native grasslands to pasture (Wilbur 1972; Studer1983). In addition, current condor populations may be too low to be affected by low habitat availability (Snyder and Schmitt 2002). However, as the wild condor population increases and expands its current foraging range, and potentially nesting site distribution, secure foraging habitat availability and safe food sources could become limiting factors for recovery of the species. Providing foraging habitat for the condor is one of the recovery objectives for the species (USFWS 1996).

Conservation and Management Activities

Since the 1980s, there has been an extensive series of conservation and management activities for the California condor, which are briefly summarized here. The reader is directed to the Recovery Plan for the California Condor (USFWS 1996) for an in-depth discussion of conservation actions prior to 1996.

In 1973, a California condor recovery team, involving the U.S. Fish and Wildlife Service (USFWS), California Department of Fish and Game (CDFG), National Audubon Society, U.S. Forest Service, Bureau of Land Management, Zoological Society of San Diego, and Los Angeles Zoo, was created and the Condor Recovery Program was initiated (USFWS 1996).

The team produced the first California Condor Recovery Plan, which was approved in 1975, with subsequent revisions in 1979, 1984, and 1996. While earlier plans focused on reducing mortality factors through habitat preservation and conservation and the initiation of a captive breeding program for California condors, the 1996 version of the plan shifted the conservation emphasis to the existing captive breeding program and reestablishment of the species in the wild (USFWS 1996).

As part of the program, all remaining individuals left in the wild were captured between 1982 and 1987 for an intensive captive bird breeding program. By 1987, a captive population of 27 individuals had been established. Captive breeding operations resulted in a substantial production in young, which prompted the initiation of a condor release program to the wild in 1992. An intensive management program, including monitoring, captive breeding, and supplemental feeding, continues to be implemented because it is needed to maintain wild populations (USFWS 2010).

Data Characterization

The California condor is one of the most thoroughly studied species in the United States. Free-flying condors have been outfitted with radiotelemetry and GPS units, and hundreds of thousands of data points have been collected. For example, the U.S. Geological Survey (USGS) 2010 study of the Southern California condor population alone analyzed 127,931 GPS locations for 21 individuals for the period of 2004 through 2009. A wealth of information and data are available for this species, and the continuing efforts at captive breeding and release ensure that this data flow will continue.

Management and Monitoring Considerations

The California condor has been one of the most managed species in the United States. As a result of this intense management, including the ongoing captive breeding program, condors have been pulled back from the brink of extinction.

Specific measures identified in the USFWS spotlight species action plan [for] 2010–2014 (2009) to reach the identified target goal of maintaining the status of the condor include the following:

1. Maintain captive reproductive rate of no less than 20 chicks per year.
2. Increase the wild populations to 280 individuals.
3. Increase yearly active breeding attempts to 35 pairs.
4. Improve annual wild nest success rates to 52%.
5. Continue monitoring for lead exposure in free-flying California condors and surrogate species and lead in the environment using carcass collection concurrent with regulation changes.
6. Continue chelation therapy treatment for all California condors with measured lead blood levels higher than 40 micrograms per deciliter.
7. Complete and publish research reports on topics related to California condor natural history, ecology, and management to be applied toward adaptive management.
8. Maintain outreach and education programs to provide information on California condor biology, ecology, and management actions.
9. Maintain outreach and education programs to provide information on non-lead alternative ammunition.

In addition, the USFWS 5-year Review included specific management and research recommendations over the next 5 years within specific programs including: priority needs, captive breeding program, field restoration activities, data analysis and management, outreach and education, and research.

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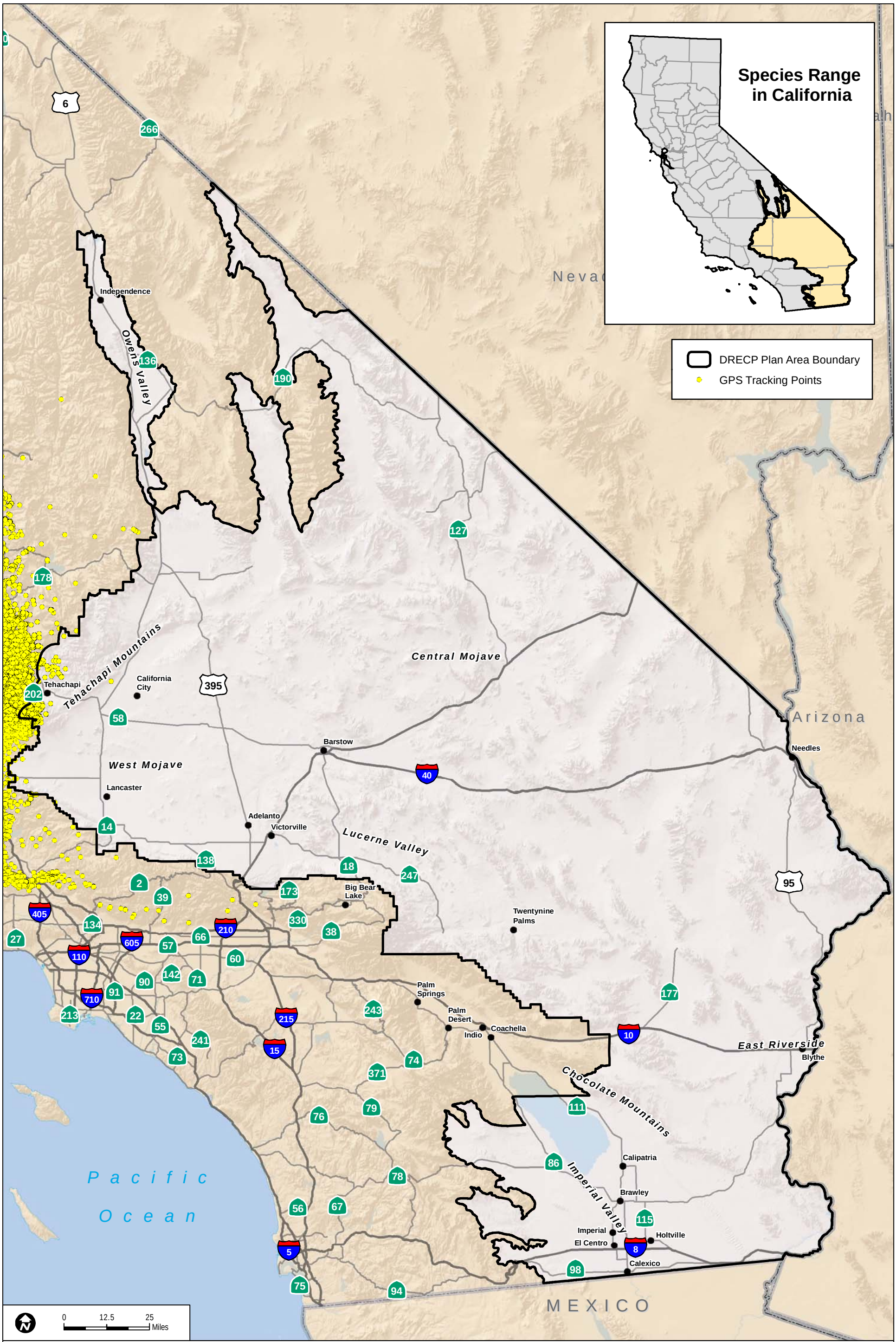

Sources: ESRI (2014); USFWS (2013), CWHR (2008)

FIGURE SP-B05

California Condor Occurrences in the Plan Area

Gila Woodpecker (*Melanerpes uropygialis*)

Photo courtesy of Dr. Lloyd Glenn Ingles, California Academy of Sciences.

Legal Status

State: Endangered
Federal: Bureau of Land Management Sensitive Species, U.S. Fish and Wildlife Service Bird of Conservation Concern
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

The Gila woodpecker (*Melanerpes uropygialis*) has been considered part of a superspecies group with red-bellied (*M. carolinus*), West Indian (*M. superciliaris*), golden-fronted (*M. aurifrons*), and Hoffmann's (*M. hoffmannii*) woodpeckers (Short 1982; AOU 1998). Peters (1948) considered it conspecific with the gray-breasted woodpecker (*M. hypopolius*), but Selander and Giller (1963) provided reasons for treating the latter as a distinct species (AOU 1998). Descriptions of the species' physical characteristics, behavior, and distribution are provided in a variety of field guides (e.g., Peterson 1990; Sibley 2000; National Geographic 2002).

Distribution

General

The Gila woodpecker's distribution ranges from near sea level in the Colorado River Valley up to 4,000 feet elevation in desert canyons and foothills (Bent 1939). The Gila woodpecker is predominantly a permanent resident across its range in areas of southeast California, southern Nevada (Alcorn 1988), central Arizona north to Mogollon Rim (Edwards and Schnell 2000), and extreme southwestern New Mexico (Hubbard 1978). It also ranges south in Mexico through Baja California, excluding northwestern Baja California Norte (Wilbur

1987) and western Mexico from the U.S.–Mexico border south to Central Mexico (Howell and Webb 1995; AOU 1998).

Distribution and Occurrences within the Plan Area

Historical

The Gila woodpecker is an uncommon to fairly common resident in Southern California along the Colorado River, and locally near Brawley, Imperial County (Garrett and Dunn 1981). Historically in southeastern California, van Rossem (1933) and Grinnell and Miller (1944) thought this species was spreading north in the Imperial Valley from the Colorado River Delta. More recently, it has declined in the Plan Area (Garrett and Dunn 1981; Rosenberg et al. 1991; Kaufman 1996). The Desert Renewable Energy Conservation Plan (DRECP) Area includes 38 historical (i.e., pre-1990) California Natural Diversity Database (CNDDB) records, all of which are along the Lower Colorado River between the area where it intersects the California state line and the Mexican border (Figure SP-B08) (CDFW 2013).

Recent

The CNDDB contains 20 recent (i.e., since 1990) occurrence locations for the Gila woodpecker in the Plan Area (CDFW 2013). All but three occur on public land (e.g., Bureau of Land Management, U.S. Fish and Wildlife Service, Bureau of Reclamation, or Imperial County); one is on private land; and two occur on land of undocumented ownership (CDFW 2013). All the recent documented occurrences in the CNDDB are along or in close proximity to the Colorado River and within the Imperial Valley, particularly south of the Salton Sea, and in desert washes as far east as Joshua Tree National Park. (Figure SP-B08). There are also 31 recent occurrences in the eBird database that mostly occur on private lands south of the Salton Sea, and one on public lands in the Lower Colorado River area (Figure SP-B08) (Dudek 2013).

Natural History

Habitat Requirements

For breeding habitat, Gila woodpeckers require cacti or trees with large trunks that are used for nesting sites. Suitable habitats include riparian woodlands, uplands with concentrations of large columnar cacti, old-growth xeric-riparian wash woodlands, urban or suburban areas, and agricultural areas (see Table 1) (Rosenberg et al. 1987; Edwards and Schnell 2000). Dominant canopy species in suitable habitat in the Plan Area include Fremont cottonwood (*Populus fremontii*) and Goodding's willow (*Salix gooddingii*) in riparian woodlands; blue palo verde (*Cercidium floridum*) and ironwood (*Olneya tesota*) in xeric-riparian woodlands; giant saguaro (*Carnegia gigantea*) in saguaro scrub communities; and various palms, eucalyptus (*Eucalyptus* spp.), and Athel tamarisk (*Tamarix aphylla*) in human-altered environments (Edwards and Schnell 2000). Rosenberg et al. (1991, 1987) found that Gila woodpeckers preferred large patches of woody riparian vegetation for nesting (greater than 49 acres), but others have documented the species in various habitat types, such as desert washes (McCreedy 2008) and residential areas (Mills et al. 1989).

Table 1. Habitat Associations for Gila Woodpecker

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Saguaro scrub	Breeding, foraging	Primary	Mature saguaro cacti for breeding (avg height = 7.8 meters [25.6 feet], > 4–5 meters [13.1–16.4 feet])	McCreedy 2008; Korol and Hutto 1984; Kerpez and Smith 1990a
Desert riparian woodland	Breeding, foraging	Primary	Mature cottonwood and willow trees	Edwards and Schnell 2000
Xeric-riparian woodland	Breeding, foraging	Secondary	For breeding, mature palo verde (avg height = 7.3 meters [23.9 feet]) or mesquite trees	McCreedy 2008; Edwards and Schnell 2000; Anderson et al. 1982

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Suburban	Breeding, foraging	Secondary	Various nonnative species, cottonwood, mesquite, and willow trees	Edwards and Schnell 2000; Rosenberg et al. 1987

Notes: avg = average; > = greater than

Foraging Requirements

Gila woodpeckers are omnivorous. They forage primarily on large trees, columnar cacti, and mistletoe (*Phoradendron californicum*), gleaning insects and eating flowers or fruit; though they will occasionally ground-feed when food is easily visible (Edwards and Schnell 2000). Seasonal patterns include feeding on saguaro and other cacti during the summer, when flowers and fruit are present, and mistletoe during the winter, when mistletoe berries are present (Edwards and Schnell 2000). Where saguaro are less common, such as the Lower Colorado River Valley, Gila woodpeckers feed primarily on insects (beetles, moths, butterflies, ants, and cicadas) (Anderson et al. 1982). In southeast California, the species has been observed as a nest predator, eating eggs of Lucy's warbler (*Vermivora luciae*), yellow warbler (*Dendroica petechia*), and Bell's vireo (*Vireo bellii*) (Edwards and Schnell 2000).

Reproduction

The breeding season throughout the Gila woodpecker's range generally begins in April and lasts through August (Anderson et al. 1982; Edwards and Schnell 2000). Fledgling occurs when nestlings are approximately 4 weeks of age (Kaufman 1996) and Gila woodpeckers will occasionally lay multiple clutches per breeding season (Phillips et al. 1964; Inouye et al. 1981). Along the lower Colorado River, fledglings appear during April (Anderson et al. 1982) and family groups with first brood offspring may remain together as adults attend to second nests (Rosenberg et al. 1991), with second broods fledgling at the end of June (Edwards and Schnell 2000). Clutch size is commonly three to five eggs (Terres 1991). For 84 egg

sets stored at the Western Foundation for Vertebrate Zoology, clutch sized ranged from two to seven eggs (mean 3.74 ± 0.87 SD) (Edwards and Schnell 2000). Both the male and female assist in incubation (Hensley 1959) and actively deliver food to young (Edwards and Schnell 2000).

Spatial Behavior

Gila woodpeckers are largely permanent local residents (Edwards and Schnell 2000). Some move short distances seasonally and, when not nesting, will move locally to concentrated food sources (Kaufman 1996).

Gila woodpecker territory size is habitat-dependent. A wash at Organ Pipe National Monument contained three territories averaging 4.6 hectares (approximately 11.3 acres) (Hensley 1954). Two territories in an "open desert area" averaged 9.9 hectares (approximately 24.4 acres) in extent (Edwards and Schnell 2000), while in a mature cottonwood stand in Grant County, New Mexico, Brenowitz (1978) observed six breeding pairs spaced 120 meters (approximately 394 feet) apart (SE ± 7 feet). Pairs defended an area up to 40 to 50 meters (approximately 131 to 164 feet) from their nest from gilded flickers (*Colaptes chrysoides*), European starlings (*Sturnus vulgaris*), and other Gila woodpeckers during the pre-nesting period of breeding season.

Ecological Relationships

Gila woodpeckers act aggressively toward numerous species, as noted in Spatial Behavior, but also provide cavities for many secondary cavity-nesters, such as the non-native European starling, which they may compete with for nest cavities (Brenowitz 1978; Kerpez and Smith 1990b). According to Brush et al (1983), in southwestern Arizona, three pairs of European starlings usurped cavities that Gila woodpeckers had used the year before (Brush et al. 1983); however, the woodpeckers excavated new cavities and bred successfully. Brenowitz (1978) observed that Gila woodpeckers were territorial toward species that overlapped with them in nest-cavity use (European starlings, gilded flickers, conspecifics) but not toward species that used different nest sites. Aggression has also been documented toward brown-crested flycatcher (*Myiarchus tyrannulus*) (Brush et al. 1983), bronzed cowbird (*Molothrus aeneus*), Bendire's

thrasher (*Toxostoma bendirei*), and curve-billed thrasher (*T. curvirostre*) by Gilman (1915), as well as toward cactus wren (*Campylorhynchus brunneicapillus*), house finch (*Carpodacus mexicanus*), and white-winged dove (*Zenaida asiatica*) by Martindale and Lamm (1984). Steenbergh and Lowe (1977) noted that Gila woodpeckers, along with several other bird species, are potentially important disseminators of saguaro cactus seeds.

Population Status and Trends

Global: Secure (NatureServe 2011)
State: Imperiled/Critically Imperiled (NatureServe 2011)
Within Plan Area: Declining (McCreedy 2008)

Recently, Gila woodpecker populations have declined significantly in southeast California (Rosenberg et al. 1991; Kaufman 1996), possibly due to the clearing of woodlands in the Colorado River Valley and Imperial Valley and nest-site competition with European starlings (Garrett and Dunn 1981). Rosenberg et al. (1991) indicated that although the species was formerly more common and widespread in Lower Colorado River Valley, it had become restricted to relatively few areas where some tall trees were retained in native habitats. About 200 breeding individuals were estimated to occur on the California side of the Lower Colorado River Valley in 1983 (Rosenberg et al. 1991), but Laymon and Halterman (1986) estimated that fewer than 30 pairs survived in California altogether. Using Breeding Bird Survey data, the Patuxent Wildlife Research Center reports a significant population trend of -2.2% (P= 0.04) for Gila woodpeckers in Arizona from 1980 to 2007, which is the time period for which most surveys have occurred (Sauer at al. 2008). McCreedy (2008) projected a negative population trend of more than 1.5% per year in southeastern California from 1966 to 2003.

Threats and Environmental Stressors

Threats and environmental stressors to Gila woodpeckers in the Plan Area include habitat loss and potentially nest site competition, with European starlings. In the southwestern United States, human development and the spread of invasive species have fragmented

and degraded riparian woodland and desert habitat, adversely affecting Gila woodpecker populations.

Water diversions, vegetation clearing for agriculture or development, grazing, recreation, wood cutting, and other human-induced disturbances have altered and fragmented riparian communities in the southwestern United States (Szaro 1989). Altered hydrology and fire regimes in the Lower Colorado River Valley have resulted in large-scale conversion of cottonwood-willow riparian forest to salt-cedar (*Tamarix* sp.) stands (Di Tomaso 1998). Gila woodpeckers will occasionally nest in large Athel tamarisk, but the more common salt-cedar stands that dominate the lower Colorado River are not viable Gila woodpecker nesting habitat (Rosenberg et al. 1991). Few mature native woodlands remain, which forces birds into less suitable habitats (Remsen 1978) and restricts the viability of local populations (Rosenberg et al. 1991). Isolated mature cottonwood-willow groves of less than 20 hectares (approximately 49.4 acres) were devoid of Gila woodpeckers in the Lower Colorado River Valley. In general, the smaller the habitat patch, the less likely it is that this species will be present (Rosenberg et al. 1991).

Human development also continues to threaten Gila woodpecker habitat in desert landscapes, facilitating invasive species spread and altering ecological processes. Invasions of several fire-adapted exotic annuals grasses have altered the fire regime in the Mojave and Colorado deserts, resulting in more extensive and frequent burns (Brooks 1999). Vegetation that Gila woodpeckers require for nesting in upland habitat, such as large columnar cacti and palo verde and mesquite trees, are not adapted to high-frequency fire regimes and thus require longer periods to recover from burns.

Vegetative species are not the only exotic species to adversely affect Gila woodpeckers. From 1968 to 1976, the number of European starlings in the southwestern U.S. more than doubled; competition between starlings and Gila woodpeckers will probably become more severe and widespread with time (Edwards and Schnell 2000, see *Ecological Relationships* for more information on nest site competition with European starlings). Furthermore, declining Gila woodpecker numbers could affect saguaro cactus populations as the woodpecker may be an important seed disperser and pollinator (Steenbergh and

Lowe 1977; Edwards and Schnell 2000). The future of this cavity-nesting bird remains highly dependent upon the continued existence of large saguaro cacti (Edwards and Schnell 2000).

Conservation and Management Activities

Large-scale cottonwood-plantation and tamarisk removal projects are underway in the Lower Colorado River Valley, which may add Gila woodpecker habitat in the future (McCreedy 2008). For example, the Lower Colorado River Multi-Species Conservation Program (LCRMSCP) has a goal of creating 1,702 acres of cottonwood-willow habitat consisting of no habitat patches less than 50 acres in size (LCRMSCP 2004); Rosenberg et al. (1991) suggest that patches of a lesser size may not support Gila woodpecker populations. However, though Gila woodpeckers are endangered in California, there are no current statewide management programs to conserve this species (McCreedy 2008).

Data Characterization

There are relatively few (16) recent occurrences in the CNDDB (CDFW 2013). No recent systematic surveys for the species have been conducted, so relatively little is known about the current population. Only Milpitas Wash has been recently surveyed in Imperial County, and the total number of breeding pairs in the county is unknown (McCreedy 2008). A census across the woodpecker's range in California, including the xeric washes in Imperial County, would inform conservation efforts as to the value of these habitats to Gila woodpecker conservation efforts. Given the extent of habitat conversion and human population growth in the Gila woodpecker's range, further investigation is warranted into the effects of human activities on the species. Although Gila woodpeckers may find certain human-dominated landscapes suitable breeding habitat (McCreedy 2008; Rosenberg et al. 1987), the species' numbers in southeastern California are still declining, warranting careful monitoring and evaluation.

Demographic data are also extremely limited for Gila woodpeckers. Studies of productivity (including data in natural versus human-dominated environments), survivorship, and fire response (e.g., nest success, emigration, carrying capacity of habitats adjacent to burns)

have not been conducted. Although competition between European starlings and Gila woodpeckers for nest cavities is documented (Kerpez and Smith 1990b; Brenowitz 1978), Koenig (2003) could not find significant evidence that European starling invasion is directly tied to Gila woodpecker population declines. This uncertainty warrants further study to determine the impact of European starlings on Gila woodpecker populations to inform conservation efforts.

Management and Monitoring Considerations

Careful monitoring of the Gila woodpecker population in the Plan Area is needed to inform conservation action implementation. There is a general lack of understanding regarding Gila woodpecker demographics in California, the most immediate or pervasive threats to the species, and its habitat preferences, including tolerance of human activities. Despite this uncertainty, large saguaro cacti or other mature trees, such as cottonwood or willow, should be given special consideration when preserving or restoring Gila woodpecker habitat. In riparian areas, woodpeckers may require more than 50 acres of woody vegetation, but Tweit and Tweit (1986) noted that residential development at a density of 2 houses per hectare (approximately 2.47 acres) did not reduce Gila woodpecker densities if native vegetation was maintained. The habitat elements that limit Gila woodpecker's use of xeric areas lacking large saguaros are not well understood, but Lynn et al. (2008) suggest that human-created water sources may be a valuable resource to resident bird populations in these environments. This study suggests that maintaining natural water sources (e.g., natural rock tanks [tinajas], springs, and ephemeral washes) in upland areas, or augmenting natural sources as necessary, is important to preserving viable Gila woodpecker habitat.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Gila woodpecker, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that

are important for species occupation, but for which information is not available for habitat modeling.

There are 1,485,338 acres of modeled suitable general habitat for Gila woodpecker in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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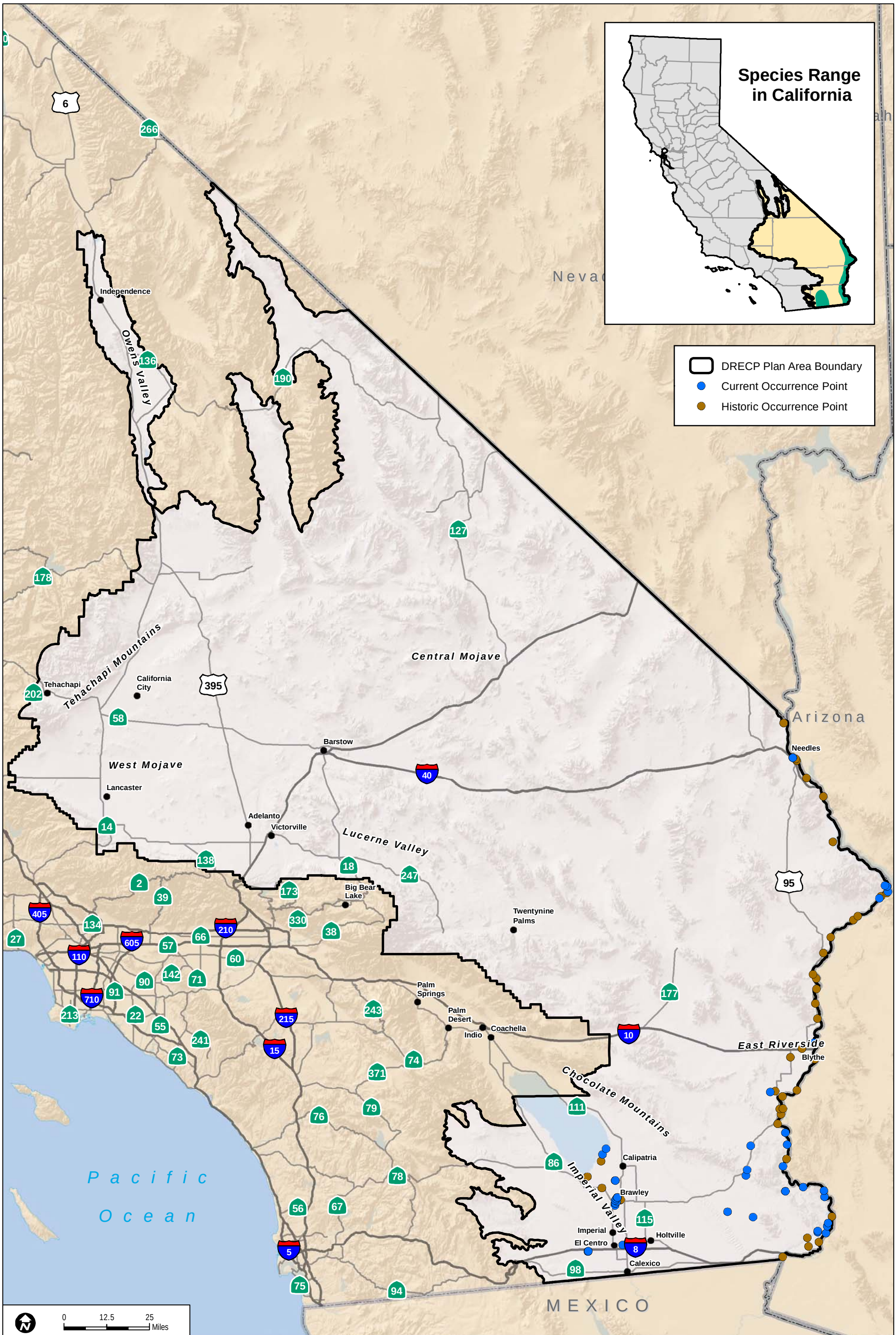

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B06

Gila Woodpecker Occurrences in the Plan Area

Golden Eagle (*Aquila chrysaetos*)

Gerald and Buff Corsi © California Academy of Sciences.

Legal Status

State: Fully Protected, Watch List
Federal: Protected under the Bald and Golden Eagle Protection Act and Migratory Bird Treaty Act, U.S. Fish and Wildlife Service Bird of Conservation Concern
Critical Habitat: N/A
Recovery Planning: N/A
Notes: Listing status not anticipated to change during permit period

Taxonomy

Of five or six golden eagle (*Aquila chrysaetos*) subspecies throughout the Northern Hemisphere, only one occurs in North America: *Aquila chrysaetos canadensis*. No information is available on geographic or genetic variation within the North American subspecies (Kochert et al. 2002).

Distribution

General

The golden eagle is predominately a western North American species, ranging from northern Alaska though the western states and Great Plains to Mexico, with some breeding and wintering locations in eastern North America (Figure SP-B09). Within California, the golden eagle is a year-round resident generally inhabiting mountainous and hilly terrain throughout the open areas of the state. Descriptions of the species' physical characteristics, behavior, and distribution are provided in a variety of field guides (e.g., Peterson 1990; Sibley 2000; National Geographic 2002).

Distribution and Occurrences within the Plan Area

Historical

The golden eagle is an uncommon permanent resident and migrant throughout the Plan Area, ranging from sea level up to 3,500 meters (11,480 feet) (Grinnell and Miller 1944). Habitat typically includes rolling foothills of oak and juniper woodlands, mountain areas, and desert. Breeding habitat is more prevalent in the southern portion of the Plan Area, including northern Imperial County, Riverside County, and southern San Bernardino County, as well as interspersed in northern San Bernardino County and the more mountainous regions of southern Inyo County (University of Washington 2011). Historically, golden eagles are rare or absent in the lower elevation desert regions of the Plan Area and the vicinity of the Salton Sea and the lower Colorado River (Kochert et al. 2002). There are 327 historical (i.e., prior to 1990) records of occurrence for golden eagle in the Plan Area and an additional 12 occurrences with an unknown observation date (CDFW 2013; Dudek 2013). There are golden eagle historical occurrences throughout the Plan Area, but with concentrations in the west Mojave, the region between Victorville and Barstow east of Interstate 15, the Mojave National Preserve, and the east portion of Joshua Tree National Park (Figure SP-B09).

Recent

There are 625 recent (i.e., since 1990) documented occurrences for golden eagle within the Plan Area (Figure SP-B09) (CDFW 2013; Dudek 2013). Golden eagles have occupied nearly every mountain range in the Plan Area; territory occupancy is variable from year to year, productivity is generally low, and most territories contain several alternate nests (La Pré 2011, pers. comm.). The Bureau of Land Management (BLM) identified "Key Raptor Areas" for golden eagles encompassing the Granite, El Paso, Newberry, and Red mountains, Stoddard Ridge, and Daggett Ridge (Raptor Research Foundation 1989). Other important occupied habitat is in the Clark Mountain Range, Tehachapi Mountains, southern Sierra Nevada Mountains, and Calico Mountains. Golden eagles may be less abundant in southeastern Imperial County (La Pré 2011, pers. comm.) Many

documented occurrences and nests exist to the southwest of the Plan Area in western Riverside and San Diego counties (CDFW 2013).

Natural History

Habitat Requirements

Golden eagles use nearly all terrestrial habitats of the western states, occurring primarily in mountainous canyon land, rimrock terrain of open desert and grassland areas (Kochert et al. 2002) (Table 1). In central California, they prefer open grasslands and oak savanna, with lesser numbers in oak woodland and open shrublands (Hunt et al. 1998) but can also be found in desert grasslands and chaparral habitats (Millsap 1981). Cliffs and large trees are used for nesting. Eagles favor cliff ledges with overhangs in areas where extreme solar radiation or high rates of precipitation threaten chick survival (Hunt, pers. comm. 2012). Preferred territory sites include those that have a favorable nest site, a dependable food supply, and broad expanses of open country for foraging (see Foraging Requirements). Hilly or mountainous country where takeoff and soaring are supported by updrafts is generally preferred to flat habitats (Johnsgard 1990). Deeply cut canyons rising to open mountain slopes and crags are ideal habitat (Kochert et al. 2002). Extensive croplands are generally avoided (Hunt, pers. comm. 2012). Golden eagles nest from 200 feet to over 9,000 feet above mean sea level (AMSL).

Table 1. Habitat Associations for Golden Eagle

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Other (rock outcrops/ barrens)	Nesting	Primary habitat	Rugged, open habitats with canyons and escarpments; secluded cliff faces with ledges extensive enough to accommodate large stick nests. Overhanging ledges preferable in extremely hot or very rainy environments.	Direct observations

Table 1. Habitat Associations for Golden Eagle

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Upland tree-dominated conifer	Nesting/ roosting, cover	Primary habitat	Large trees, near suitable ground squirrel and other prey habitat; trees large enough to support the large nest structure (up to 3 meters across and 1 meter deep)	Direct observations and radiotelemetry studies
Grasslands	Foraging	Secondary habitat	Relatively open and expansive rolling foothills and mountain terrain, often with wide plateaus cut by streams or canyons on open mountain slopes	Direct observations and radiotelemetry studies, and aerial surveys

Sources: Kochert et al. 2002, Hunt, pers. comm. 2012
m – meter

Foraging Requirements

Golden eagles typically forage in open habitats including grasslands and shrublands. They feed mainly on leporids (hares and rabbits) and sciurids (ground squirrels, prairie dogs, marmots), but they also take birds, fish, and reptiles, and frequently feed on carrion (Kochert et al. 2002). Hunting strategies are variable and include attack glides from soaring flight, low-level glides over open hilly terrain ("contour hunting"), and attacks from a perch (Kochert et al. 2002; Polite and Pratt 1990). Golden eagles often pirate food from other raptors. Hunting in mated pairs is also documented (Kochert et al. 2002).

Reproduction

Golden eagles attain adult plumage in their fifth summer (Kochert et al. 2002). In healthy populations, many adults are prevented from obtaining a breeding territory until a vacancy arises through the death

of an established pair member (Haller 1996). These unmated adults ("floaters") form a reserve of potential breeders that buffer the breeding population against loss (Hunt 1998). High mortality, particularly among the older age categories, may reduce or eliminate the floater buffer and cause the overall population to decline.

Mated pairs may use the same nest each year, or use alternate nests within their territories (Terres 1991). Pairs rarely re-nest when the first clutch is destroyed (Watson 1997) and there are no records of pairs producing more than one brood per year. Golden eagles prefer to locate their nests on cliffs or in trees near forest edges or in small stands near open fields (Bruce et al. 1982; Hunt et al. 1998). Breeding densities are directly related to territorial spacing and foraging requirements for the species. The breeding cycle extends from late January through August, with peak activity from February through June. Eggs are laid from early February to mid-May (February and March in most of California). Clutch size varies from one to four eggs, but two is the most common size (Brown 1976; Johnsgard 1990). Incubation lasts 43–45 days (Kochert et al. 2002), and the fledging period is 72–84 days (Johnsgard 1990). The young usually remain dependent on their parents for as long as eleven weeks after fledging. Long-term annual reproductive success (number of large young per occupied territory) ranges from 0.64 to 1.08 fledglings per pair in the continental United States, varying with prey abundance and weather (Phillips et al. 1990; Thompson et al. 1982).

Table 2. Key Seasonal Periods for Golden Eagle

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding	✓	✓	✓	✓	✓	✓	✓	✓				
Migration		✓	✓	✓					✓	✓	✓	✓

Notes: Active year-round residents in Southern California.

Sources: Polite and Pratt 1990; Kochert et al. 2002

Activity and Movement

Golden eagles in the Plan Area are mostly resident (Polite and Pratt 1990). Dixon (1937) estimated an average home range size of about 93 kilometers^2 (36 miles^2) in Southern California, but home range can vary substantially with habitat conditions and prey availability. In the western U.S., on average, eagles forage over home ranges ranging from about 22 to 33 kilometers^2 (8.5 to 12.7 miles^2) during the breeding season (Kochert et al. 2002). Resident pairs maintain home ranges year-round with shifts in intensity of use from the breeding season to winter (Dunstan et al. 1978; Marzluff et al. 1997). Both residents and migratory individuals show fidelity to wintering areas (Kochert et al. 2002). Though limited dispersal data exist, three radio tagged resident breeders in California all moved to new territories within 8 kilometers after leaving their original ones (Kochert et al. 2002). Some migrants may temporarily move into areas used by resident birds during the winter.

Table 3. Spatial Behavior of the Golden Eagle

Type	Distance/Area	Location of Study	Citation
Territory	93 km^2	Southern California	Dixon 1937
Distance between active nest sites	>0.8 km	Western United States	Palmer 1988
Dispersal from natal site	Ranged from 6.7 to 64.7 km	Idaho	Steenhof et al. 1984

> – greater than
km – kilometer

Ecological Relationships

Golden eagles are a top avian predator in the scrubland, grassland, and woodland ecosystems that make up much of the Plan Area. They may directly compete with ferruginous hawks (*Buteo regalis*) and other hawks for mammal prey, and with California condors (*Gymnogyps californianus*), turkey vultures (*Cathartes aura*) and ravens (*Corvus corax*) for carrion. Territorial interactions with other golden eagles may result in some fatalities.

Population Status and Trends

Global: Secure (NatureServe 2011)
State: Vulnerable (NatureServe 2011)
Within Plan Area: Apparently stable (Remsen 1978)

The golden eagle is relatively common in some areas of its range. Local threats or declines do not currently pose a major conservation problem from a global perspective (NatureServe 2011). This species was once a common resident throughout the open areas of California. Numbers are now reduced near human population centers; nesting populations in San Diego County decreased from an estimated 85 pairs in 1900 to 40 occupied territories in 1999 due to extensive residential development (Kochert et al. 2002).

Threats and Environmental Stressors

Golden eagle declines, where they have occurred, are attributed primarily to habitat degradation and human-induced disturbances and mortality (Kochert et al. 2002). Golden eagles are particularly sensitive to human activity near nests, especially during incubation and before the young can thermoregulate (at approximately 3 weeks or age). Golden eagles may be secondarily poisoned by consuming prey that has itself been poisoned by chemicals used to protect crops or kill rodents (Kochert et al. 2002). Additional mortality agents are poaching, electrocution from distribution and utility lines, wire strikes, wind turbine strikes, and lead poisoning (Remsen 1978; Thelander 1974). In a study of the causes of fatalities in 61 golden eagles radio-tagged and recovered in the Diablo Range from January 1994 to December 1997, 37% were killed by wind turbine strikes, 16% by electrocution, and 5% by lead poisoning (Hunt et al. 1998); additional poisoning deaths were suspected in undiagnosed fatalities not involving trauma. The pervasiveness of lead in the environment in the remains of gun-killed animals may impact golden eagle populations. Evidence of elevated blood-lead levels (greater than 0.20 parts per million), likely from ingested hunter ammunition, was detected in 36% of 162 eagles from Southern California from 1985 to 1986 (Harlow and Bloom 1989; Pattee et al. 1990). More than 270 eagles were electrocuted in North America during 1986-1996 (Harness and Wilson 2001); ieagles are most susceptible to

electrocution when landing on power poles where parallel wires are close together (Kochert et al. 2002). Vehicle collisions have also been documented as a cause of mortality (Phillips 1986). Studies have documented heat stress as a significant mortality factor for nestlings (Mosher and White 1976), and an inverse correlation exists between nesting success and the number of days with temperatures greater than 32°C (89.6°F) (Steenhof et al. 1997).

Conservation and Management Activities

There are no conservation actions in the Plan Area directed specifically at the golden eagle. However, land preservation in the Southern California desert and surrounding areas by agencies such as the National Park Service, Bureau of Land Management, Department of Defense, and California State Parks have indirectly benefited golden eagles by preserving open space. Management practices on these lands that enhance golden eagles' prey base (e.g., rodents, hares, and rabbits), would likely confer additional benefits. Furthermore, the Bureau of Land Management identifies the golden eagle as a sensitive species within the Plan Area (BLM 2007). Golden eagle management and conservation generally includes habitat management, hazard management, education, and controlling human activity in sensitive raptor areas, especially during the nesting season.

The USFWS released a *Draft Eagle Conservation Guidance* document in January 2011 (USFWS 2011). This document provides guidance for preparation of Eagle Conservation Plans (ECPs) related to wind energy facilities. It would be a voluntary program for project proponents, but they would have to coordinate with the USFWS if a different approach were taken to ensure that alternative approaches would provide comparable data (USFWS 2011). The evaluation of a proposed wind energy project would be conducted in five stages:

1. *Stage 1: Identify potential wind facility locations with manageable risk to eagles at the landscape level.*
2. *Stage 2: Obtain site-specific data to predict eagle fatality rates and disturbance take at wind facility sites that pass Stage 1 assessment.*

3. *Conduct turbine-based risk assessment and estimate the fatality rate of eagles for the facility evaluated in Stage 2, excluding possible advanced conservation practices (ACPs).*
4. *Identify and evaluate ACPs that might avoid or minimize fatalities identified in Stage 3. When required to do so, identify compensatory mitigation necessary to reduce any remaining fatality effect to a no-net-loss standard.*
5. *Document annual eagle fatality rate and disturbance effects. Identify additional ACPs to reduce observed level of mortality, and determine if initial ACPs are working and should be continued. When appropriate, monitor effectiveness of compensatory mitigation.*

(USFWS 2011, p. 6).

At the end of each of the first four stages, the project proponent would determine which of the following categories the project, as planned, would fall into: (1) high risk to eagles, little opportunity to minimize effects; (2) high to moderate risk to eagles, but with an opportunity to minimize effects; (3) minimal risk to eagles; or (4) uncertain. The USFWS recommends that projects that fall into category 1 be moved, significantly redesigned, or abandoned because they likely would not meet the regulatory requirement for an ECP and permit issuance. Projects that fall into categories 2, 3, and 4 would be candidates for an ECP and permit (USFWS 2011).

The Draft Eagle Conservation Guidance is currently under review and has not been formally adopted by the USFWS.

Data Characterization

Several regional surveys in portions of the Plan Area for golden eagle have been conducted by the Wildlife Research Institute, Inc. (WRI), including an area of approximately 4,142 kilometers^2 (1,600 miles$^{2)}$ in the eastern Mojave Desert in San Bernardino and Riverside counties (WRI 2010), in the western Mojave Desert (WRI 2002), on BLM Open Areas in the Johnson and Stoddard valleys (WRI 2003, 2009a), and in Anza Borrego State Park (WRI 2009b). These studies have collected data for golden eagle nests and alternative nests, including appraisals

of nest condition, whether active or not, nest elevation, GPS coordinates, nest substrate (cliff, transmission tower, etc.), breeder age class, and behavior (e.g., WRI 2010). In addition, annual nesting surveys in San Diego County have been conducted since 1988, including the desert regions of eastern San Diego County (Unitt 2004). In other areas of California, extensive long-term studies have been conducted in the central coast ranges of California on the distribution, demographics, and general biology of golden eagles as part of investigations on the impact of wind turbine operation on this species (Hunt et al. 1998). These studies provide detailed information on the distribution and habitat-use patterns of resident and nonresident golden eagles, population structure, reproductive rates, survival rates, and population equilibrium dynamics in the central coast ranges of California. Some additional literature, some of which pertains to Southern California, is available for the golden eagle because it is a highly visible, fully protected bird of prey and a top avian predator within its range. Most of the literature pertains to general natural history, behavior, distribution, and population changes in the past 30 to 40 years. Some information is available on demographics and population trends. Limited species-specific management information is available.

Management and Monitoring Considerations

Management of healthy eagle populations includes maintaining prey habitat in foraging areas by maintaining native grassland, shrub, or woodland communities depending on foraging and nesting relationships (Marzluff et al. 1997; Kochert et al. 1999), protecting foraging habitat within 3 kilometers (1.9 miles) of nests from human disturbance and fire, and restoring shrubs in burned areas (Kochert et al. 1999). Fires have caused large-scale losses of shrubs and degraded prey (e.g., rabbit) habitat in areas used by eagles throughout California. Thus recovery of these areas as foraging habitat is important.

As discussed above under Threats and Environmental Stressors, human activities near nests can cause nest failure and nest abandonment. Planned activities in the Plan Area should consider what management actions and monitoring considerations are required to avoid and minimize human impacts to nest sites, including seasonal restrictions on certain activities near active nests and

protective buffer zones (both spatial and visual) around active nest sites. Monitoring of nest sites in areas where human activities are occurring would help distinguish between relatively benign activities that are tolerated by golden eagles and activities that disturb birds.

Another important consideration for management and monitoring of golden eagle populations is ensuring that eagles have access to safe food sources. Agricultural activities, for example, may affect golden eagles through contamination of prey by chemicals used to protect crops, including phorate, carbofuran, strychnine, and anticoagulant rodenticides (Kochert et al. 2002).

Other human-caused sources of mortality for golden eagles that may warrant monitoring and management and/or design specifications to minimize threats include wind turbine and vehicle collisions (Hunt et al. 1998; Phillips 1986) and electrocutions from power lines (Harness and Wilson 2001). Utility companies such as Southern California Edison incorporate anti-perching and anti-collision guidelines in design of transmission line facilities consistent with the Avian Power Line Interaction Committee (APLIC 2006).

As discussed above, the USFWS recently released the Draft Eagle Conservation Guidance for public review (USFWS 2011). The Stage 5 objective is annual monitoring of eagle mortality and disturbance effects, the effectiveness of compensatory mitigation, and identification of additional advanced conservation practices (ACPs) to reduce mortality and other adverse effects (USFWS 2011).

Development of a population monitoring strategy should be a priority, especially in the western United States where population declines are suspected (Kochert et al. 2002)

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for golden eagle, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for

species occupation, but for which information is not available for habitat modeling.

There are 11,219,198 acres of modeled suitable habitat for golden eagle in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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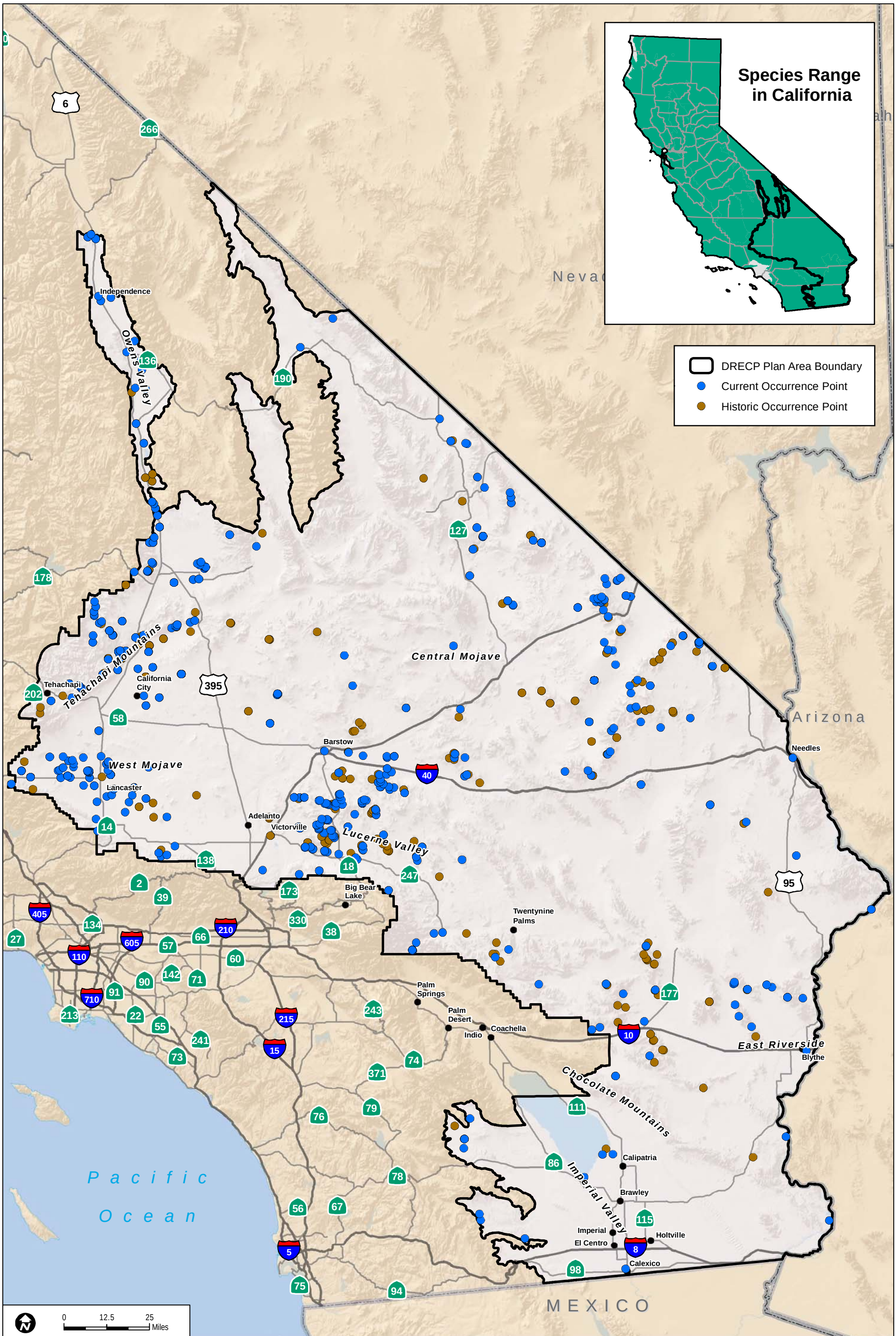

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B07

Golden Eagle Occurrences in the Plan Area

Desert Pupfish (*Cyprinodon macularius*)

Photo courtesy of Sharon Keeney, CDFW

Legal Status

State: Endangered
Federal: Endangered
Critical Habitat: 51 FR 10842–10851
Recovery Planning: Desert Pupfish Recovery Plan (USFWS 1993)

Taxonomy

The desert pupfish complex was historically comprised of two subspecies, the nominal desert pupfish (*Cyprinodon macularius macularius*) and the Quitobaquito pupfish (*Cyprinodon macularius eremus*), and an undescribed species, the Monkey Spring pupfish (*Cyprinodon* sp.) (USFWS 1993). The subspecies are now recognized as three separate species (USWFS 2010): the desert pupfish (*C. macularius*), the Sonoyta (Quitobaquito) pupfish (*C. eremus*) (Echelle et al. 2000), and the undescribed Monkey Springs pupfish, which has since been described and renamed the Santa Cruz pupfish (*C. arcuatus*). Recent work (Echelle et al. 2007; Koike et al. 2008) and a summary by the U.S. Fish and Wildlife Service (USFWS 2010) provide the evidence that *C. macularius* and *C. eremus* are separate species. The Sonoyta pupfish persists in only two populations: one near the U.S.–Mexico border at Quitobaquito Springs in Organ Pipe Cactus National Monument in Arizona, and the other at Rio Sonoyta in Sonora, Mexico (USFWS 2010). The Santa Cruz pupfish occurred in the upper Santa Cruz River basin in southern Arizona and Northern Sonora, Mexico. It is now extinct due to habitat alteration and introduced fishes (Minckley et al. 2002). All other populations are referred to *C. macularius*. Descriptions of the species' physical characteristics can be found in USFWS (1993, 2010).

Distribution

General

The desert pupfish occurs in desert springs, marshes, and tributary streams of the lower Gila and Colorado River drainages in Arizona, California, and Mexico. Natural populations of desert pupfish also occur in the Salton Sea and associated irrigation drains and shoreline pools. It also formerly occurred in the slow-moving reaches of some large rivers, including the Colorado, Gila, San Pedro, and Santa Cruz.

Distribution and Occurrences within the Plan Area

Historical

Historically, desert pupfish occurred in the lower Colorado River in Arizona and California, from about Needles downstream to the Gulf of Mexico and onto its delta in Sonora and Baja (CVAG 2007). In California, pupfish inhabited springs, seeps, and slow-moving streams in the Salton Sink basin, and backwaters and sloughs along the Colorado River. Desert pupfish also occurred in the Gila River Basin in Arizona and Sonora, including the Gila, Santa Cruz, San Pedro, and Salt Rivers; the Rio Sonoyta of Arizona and Sonora; Puerto Penasco, Sonora; and the Laguna Salada Basin of Baja California.

Recent

Because *C. eremus* occurs only in southern Arizona and Mexico (USFWS 2010) and *C. arcuatus* is now extinct, their distribution information is not discussed further; *C. macularius* is described within the Plan Area (see Figure SP-F01). USFWS (2010) describes that currently five natural populations persist in California, restricted to two streams tributary to, and many shoreline pools and irrigation drains of, the Salton Sea: San Felipe Creek/San Sebastian Marsh, Salt Creek (within the Dos Palmas Conservation Area of the Coachella Valley Multiple Species Habitat Conservation Plan [MSHCP; CVAG 2007]), Salton Sea, irrigation drains of the Salton Sea, and a wash near Hot Mineral Spa (a natural population added since the 1993 recovery plan). The desert pupfish population in Salt Creek is stable to increasing, and currently has few non-native species (Keeney 2010a,

cited in USFWS 2010). San Felipe Creek also has a stable to increasing population. California Department of Fish and Wildlife (CDFW) surveys have found a persistent population of western mosquitofish (*Gambusia affinis*) in San Felipe Creek in recent years. In addition, there are a number of refuge or captive populations of desert pupfish in California at a variety of sites (USFWS 2010): Anza-Borrego State Park; Oasis Springs Ecological Reserve; Salton Sea State Recreation Area; Dos Palmas Reserve; Living Desert Museum; University of California, Riverside; and Borrego Springs High School. The Coachella Valley MSHCP (CVAG 2007) also describes a refuge population in the larger pools around the Thousand Palms oasis area where restoration is in progress. There are no pupfish currently present here, but there are plans to restock this site when restoration has been completed.

Natural History

Habitat Requirements

Found in water of desert springs, small streams, and marshes below 1,515 meters (5,000 feet) elevation (USFWS 1993), this species tolerates high salinities, high water temperatures, and low dissolved-oxygen concentrations. In the mid-2000s CDFW found desert pupfish in the Salton Sea at depths of 7 to 8 feet while conducting fish monitoring surveys. Pupfish typically prefer clear water, with either rooted or unattached aquatic plants, restricted surface flow, and sand–silt substrates (Black 1980; USFWS 1993). Pupfish use shallow water habitats extensively, often occupying such habitat at temperatures that are above the thermal optimum for invasive fishes. Pupfish do well if these habitats have little vegetation apart from mats of benthic algae over a fine-grained mineral or detrital substrate; they also utilize areas with aquatic or emergent vascular vegetation (ICF 2009). Desert pupfish in general are noted for their tolerance of environmental stress; they can tolerate dissolved-oxygen concentrations as low as 0.13 parts per million (Helfman et al. 1997). Their temperature tolerance ranges from a low of 4.4°C (Schoenherr 1990) to a high of 42.4°C (Carveth et al. 2006). Their salinity tolerance ranges from 0 to 70 parts per thousand for eggs and adults (Barlow 1958; Schoenherr 1988) and up to 90 parts per thousand for larvae (Schoenherr 1988). Martin

and Saiki (2005) found that desert pupfish abundance was higher when vegetative cover, pH, and salinity were high and when sediment factor and dissolved oxygen were low. They hypothesize that water quality extremes (especially high pH and salinity, and low dissolved oxygen) limit the occurrence of nonnative fishes.

Table 1. Habitat Associations for Desert Pupfish

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Water of desert springs, small streams, and marshes	Breeding/ foraging	Primary habitat	Clear water, with either rooted or unattached aquatic plants, restricted surface flow, and sand–silt substrates	Direct observational studies

Sources: Black 1980; USFWS 1993; Martin and Saiki 2005.

Foraging Requirements

Pupfish are opportunistic omnivores, thriving on a diet of algae, aquatic plants, detritus, and small invertebrates (Sutton 1999, citing Crear and Haydock 1971 and Naiman 1979). Adult foods include ostracods, copepods, and other crustaceans and insects; pile worms; mollusks; and bits of aquatic macrophytes torn from available tissues (USFWS 1993). Legner et al. (1975) found that desert pupfish were more effective than mosquitofish at controlling mosquito populations. Pupfish have also been known to eat their own eggs and young on occasion. Detritus or algae are often predominant in their diets (USFWS 1993). Pit digging, the active excavation of soft bottoms in search of food, is a pupfish behavior described by Minckley and Arnold (1969); these pits are defended when occupied. Foraging is typically a daytime activity, and fish may move in response to daily warming from shallower water during morning to feed in deeper places later in the day (USFWS 1993).

Reproduction

Desert pupfish may become sexually mature as early as 6 weeks of age at 1.5 centimeters in length under conditions of abundant food and suitable temperature. Desert pupfish typically live for a year, but may live as long as 2 to 3 years. Although they may breed during their first summer, most do not breed until their second summer, when their length may have reached a maximum of 7.5 centimeters (Moyle 2002). In favorable conditions a pair of pupfish can produce 800 eggs in a season (ICF 2009). Eggs appear to be randomly deposited within the male territory. Although males actively patrol and defend individual territories, there is no directed parental care (USFWS 1993).

Table 2. Key Seasonal Periods for Desert Pupfish

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding			X	X	X	X	X	X	X			

Source: USFWS 1993.

Spatial Behavior

McMahon and Tash (1988) found that when desert pupfish occupied open pools, 84% of the total number produced emigrated. They found that when pupfish were prevented from emigrating, pupfish exhibited symptoms of overpopulation. Characteristics of overpopulation were not apparent in pupfish occupying open pools. Seasonal temperatures influenced the timing and magnitude of emigration. In summary, pupfish may regulate their populations via emigration.

Many of the locations where they are currently found are isolated from other populations. However, complete isolation mainly has been an issue in artificial populations, although even in these populations "complete isolation" no longer occurs given CDFW's recent inoculation of refuges with wild fish. Most natural populations have some connection to other populations occasionally (e.g., via flash flood), although these opportunities for mixing are brief and infrequent. This may become more of an issue given the uncertainty of the Salton Sea.

Desert pupfish congregate in the summer where adult females swim in loose schools and leave the school when attracted by a territorial male to spawn. Pupfish movement between the Salton Sea and nearby drains has been observed (Sutton 1999). Sutton (2002) describes desert pupfish summer movement between a drain (although not connected directly to the Salton Sea) and a shoreline pool, as well as movement of approximately 0.5 kilometer (0.3 mile) from Salt Creek to a downstream shoreline pool (although not connected to the Salton Sea). Sutton (2002) hypothesizes that movements from Salt Creek to the shoreline pool were due to water level drops. The technique used by Sutton (2002) for tracking desert pupfish holds promise for further desert pupfish movement studies.

Table 3. Spatial Behavior by Desert Pupfish

Type	Distance/Area	Location of Study	Citation
Breeding territory	Normally defends 1 to 2 square meters but as large as 5 to 6 square meters	Not disclosed	Moyle 1976

Ecological Relationships

The desert pupfish were once found in varying water bodies from cienegas and springs to shallow streams and margins of larger bodies of water where they preferred shallow, slower-moving water with soft substrates and clear water (USFWS 1993). Over the last century, land use activities such as groundwater pumping, dewatering, water diversion, and drain maintenance have altered the water levels, resulting in habitat loss for desert pupfish. Channel erosion can increase the sediment in the water, reducing its suitability for the pupfish; water impoundment creates deeper ponds that increase occupation by non-native aquatic species; and grazing practices reduce vegetative cover, increase sedimentation, and trample habitat (USFWS 1993). Off-road vehicle use can be problematic in some areas, and currently is more of an issue than is grazing.

Currently, the major threat to the species is the presence of exotic aquatic species, particularly tilapia (*Tilapia* spp.), sailfin molly (*Poecilia latipinna*), western mosquitofish, several snail species, and crayfish (*Procambarus clarkii*). These and other introduced fish species primarily affect pupfish populations through predation, competition, and behavioral interference (CVAG 2007). Introduced fishes (and other aquatic organisms) can affect pupfish populations via other means as well, such as disease and habitat displacement. Additionally, in a few areas, such as San Felipe Creek and Salt Creek, where non-native fishes are relatively few (at least currently), the most serious threat may be the abundance of tamarisk/salt cedar (*Tamarix* spp.).

The desert pupfish appears to go through cycles of expansion and contraction in response to natural weather patterns (51 FR 10842–10851; USFWS 1993; Weedman and Young 1997, cited in USFWS 2010). In very wet years, populations can rapidly expand into new habitats (Hendrickson and Varela-Romero 1989, cited in USFWS 2010). In historical times, this scenario would have led to panmixia among populations over a very large geographic area (USFWS 1993).

Population Status and Trends

Global: Critically imperiled (NatureServe 2011)
State: Same as above
Within Plan Area: Same as above

In its 5-year review, USFWS (2010) concluded that threats to the species and their overall level of intensity remain similar to when the species was originally given a recovery priority number of 2C. Priority number 2C is indicative of a high degree of threat, a high potential for recovery, and taxonomic classification as a species.

Threats and Environmental Stressors

USFWS (2010) summarizes the threats to desert pupfish survival. These include threats relating to destruction or curtailment of habitat or range (USFWS Factor A), including loss and degradation of suitable habitat through groundwater pumping or water diversion; contamination from agricultural return flows, as well as other

contaminants; and physical changes to water properties involving suitable water quality. There is no new information to suggest that overutilization for commercial, recreational, scientific, or educational purposes (USFWS Factor B) are threats. The effect of disease or predation (USFWS Factor C) is a potential threat to desert pupfish. Currently, the specific effects to individual desert pupfish or populations from disease or parasites are unknown. Predators and competitors of the desert pupfish include tilapia, sailfin mollies, shortfin mollies (*Poecilia mexicana*), mosquitofish, porthole livebearers (*Poeciliposis gracilis*), and several members of the families *Centrarchidae*, *Ictaluridae*, and *Cyprinidae*, as well as melianias (*Melanoides tuberculata* and *Tarebia granifera*), crayfish, Rio Grande leopard frog (*Lithobates berlandieri*), and bullfrog (*Rana catesbeiana*) (51 FR 10842–10851; Black 1980; ICF 2009). Invasive snails (melianias) consume the algal mats that form the pupfish's principal food source (ICF 2009). They also may cause disease. For example, red-rim melania (*Melanoides tuberculatus*) is a host of parasites, including gill trematode. Known fish hosts of the gill trematode include Comanche Springs pupfish (*Cyprinodon elegans*). Juvenile tilapias compete with desert pupfish for many of the same food items (Matsui 1981); and crayfish, frogs and adult tilapia prey on fish and fish eggs (51 FR 10842–10851; ICF 2009; Matsui 1981). Crayfish were thought to be responsible for elimination of the Owens pupfish, *C. radiosus*, from a refuge in Warm Springs near Big Pine, California (Black 1980). Additionally non-native crayfishes are well known to negatively affect water quality and severely reduce, if not eliminate, algae that is favored by pupfish. These and other introduced aquatic species affect pupfish populations through predation, competition, and behavioral interference. Inadequacy of existing regulatory mechanisms (USFWS Factor D) is a potential threat to desert pupfish. Regulatory mechanisms exist in much the same state as at the time of listing, though the application of recent case law may result in reduced consideration of impacts to isolated waters containing desert pupfish (USFWS 2010). Finally, other natural or manmade factors affecting the continued existence of desert pupfish (USFWS Factor E) have been noted as a threat for desert pupfish (USFWS 1993). The only new threat identified is endocrine disruptors noted in the Salton Sea irrigation drains (USFWS 2010).

Conservation and Management Activities

The Coachella Valley MSHCP (CVAG 2007) lists some conservation and management actions that would benefit pupfish:

1. Complete hydrologic studies for the Salt Creek area to determine if the water sources for Salt Creek are adequately protected or if additional water sources may be needed and are available.[1]
2. Ensure persistence of pupfish populations in agricultural drains by managing agricultural drain maintenance and water supply. Monitoring will include surveys for pupfish presence in the agricultural drains along with regular sampling of flow, water depth, and selenium concentrations
3. Control and manage exotic or invasive species in pupfish habitat, if monitoring identifies this as a threat. Control efforts should address nonnative fish, bullfrogs, and other invasive species. The presence and potential impacts of Asian tapeworm, a potential pupfish parasite, shall also be addressed.
    a. Remove tamarisk (salt cedar) where it is affecting the amount of water available to pupfish.
4. Maintain water levels, water quality, and proper functioning condition of ponds, springs, and drains, to the extent these activities are under Plan authority, which will include reevaluating the feasibility of available technologies to reduce selenium concentrations.
5. Restore and enhance degraded habitat as necessary according to monitoring results.
6. Conduct experiments on the timing and mechanics of drain cleaning that would minimize impacts to desert pupfish.
7. Estimate distribution and/or population size of desert pupfish.
8. Survey contaminant levels in the water and in pupfish.

[1] San Felipe Creek and associated wetlands are not within the Coachella Valley MSHCP area, but complete hydrologic studies are needed for this system as well. This will be particularly important given potential impacts of climate change.

USFWS (2010) also lists some general future conservation and management activities:

- A specific standardized genetic protocol should be developed, using work by Echelle et al. (2007), as a template for management of *C. macularius* refuge populations. CDFW is currently working on this issue as part of the Desert Pupfish Refuge Management Plan being developed to provide guidance for the management of pupfish refuges (artificial habitats). Their recommendations include establishing large primary refuge populations, with each one representing the groups of wild *C. macularius*. They also recommend that secondary refuges representing each of the wild source regions be established.
- A recovery plan amendment or revision should be made based on recommendations by Loftis et al. (2009) that delineate a different set of management units in the Salton Sea than is recognized in the existing recovery plan and to reflect the changed taxonomy.
- Conservation at wild sites should be given the highest priority.
- A Safe Harbor Agreement or similar tool for the desert pupfish in California should be pursued.

Additionally, another desired study is determining the tolerance of pupfish eggs to desiccation; this study is currently being planned and is expected to occur soon.

Data Characterization

Loftis et al. (2009) assessed the mitochondrial DNA (mtDNA) results from the 1997 and 1998 surveys by Echelle et al. (2000) and used data from 10 microsatellite DNA loci to describe the genetic structure of the two extant species (*C.* macularius and *C. eremus*). According to Loftis et al., this data showed that there "was evidence ($R_{ST}>F_{ST}$) that the two extant populations of *C. eremus* have been isolated sufficiently long for mutation to contribute significantly to genetic divergence, whereas divergence among the nine assayed populations of *C. macularius* could be attributed to genetic drift alone." The assessment suggests that based on variability among the mtDNA, there are two

populations of *C. eremus* and five groups of populations of *C. macularius* that should be managed as units for conservation genetics management of the two species.

The distribution of the species and principal threats to its continued existence are sufficiently well known to allow coverage of this species in the Desert Renewable Energy Conservation Plan.

Management and Monitoring Considerations

As summarized above, the Coachella Valley MSHCP (CVAG 2007) lists some specific conservation and management actions for the Plan Area that would benefit pupfish. In addition, invasive species management options for the Dos Palmas Area of Critical Environmental Concern have been prepared (ICF 2009) and cover threats to the desert pupfish. Within that document, specific management actions that may be used to eliminate non-native aquatic species or create predator-free environments are evaluated; these include water management that alternately inundates and desiccates habitat, creation of channel habitat, creation of shallow-water habitat, removal and/or burning of emergent aquatic habitat, and invasive aquatic species trapping. As mentioned previously, CDFW is preparing the Desert Pupfish Refuge Management Plan, which will address specific management issues including control of aquatic fauna and flora, genetic protocols for monitoring of pupfish, management recommendations for each refuge, pupfish population monitoring, and other topics. The Desert Pupfish Recovery Plan (USFWS 1993) emphasizes securing extant wild populations of desert pupfish to preserve original genetic material, and creating a second and third tier of populations from these existing wild populations using a genetic exchange protocol that would be created to mimic desert pupfish evolution. Refuge population or new habitat may not be difficult to create as is evidenced by the shallow-water habitat that was constructed near the Alamo River, which was designed to exclude fish, but desert pupfish got into the ponds and flourished (Roberts 2010, cited in USFWS 2010; Saiki et al. 2011). However, habitat may be difficult to maintain in terms of costs. Bureau of

Reclamation spent three million dollars constructing, operating and maintaining this habitat before running out of funding.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for desert pupfish, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 8,155 acres of modeled suitable habitat for desert pupfish in the Plan Area. A figure showing the modeled suitable habitat in the Plan Area are included in Appendix C.

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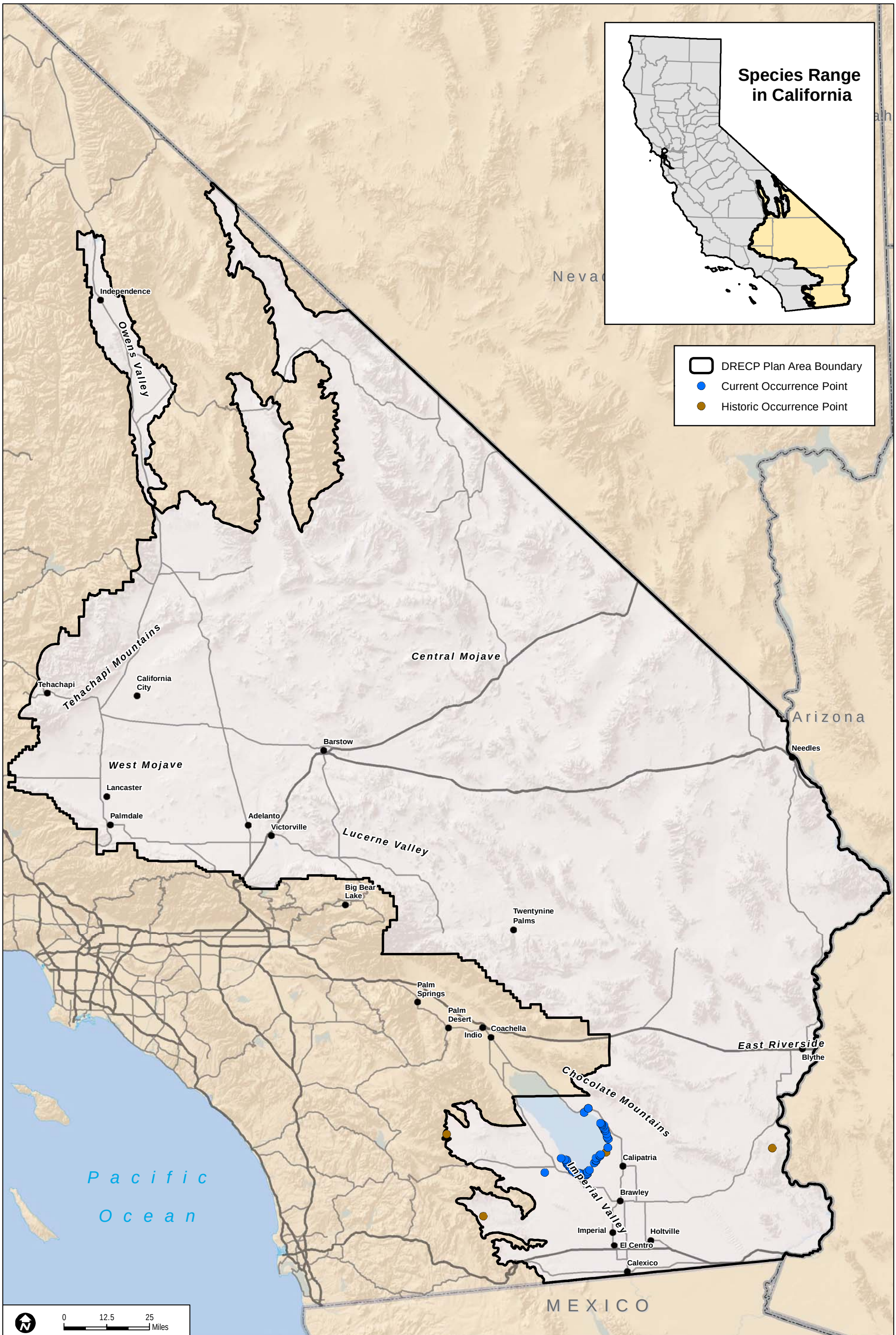

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-F01

Desert Pupfish Occurrences in the Plan Area

Mohave Tui Chub (*Siphateles bicolor mohavensis*)

Legal Status

State: Endangered, Fully Protected
Federal: Endangered
Critical Habitat: N/A
Recovery Planning: Recovery Plan for the Mohave Tui Chub, *Gila bicolor mohavensis* (USFWS 1984)
Notes: California Department of Fish and Wildlife (CDFW) has adopted the genus *Siphateles* for the species, which was previously classified under the genus *Gila.*

Taxonomy

The Mohave tui chub (*Siphateles bicolor mohavensis*) is recognized as the only fish native to the Mojave River basin in San Bernardino County. It is a member of the minnow family (Cyprinidae). It was originally identified as *Algansea formosa* in 1857 by Girard, but in 1918 Snyder described it as a new species, *Siphateles mohavensis* (as cited in USFWS 1984). Miller (1961) and Bailey and Uyeno (1964) relegated the subgenus *Siphateles* to the genus *Gila*, and in 1973 Miller reclassified the Mohave tui chub to the subspecies *G. b. mohavensis* (as cited in USFWS 1984). Simons and Mayden (1998) published a paper addressing the classification of the North America genera of Cyprinidae and, based on ribonucleic acid (RNA) sequences, restored *Siphateles* from a subgenus to a full genus. The CDFW currently includes the species under the genus *Siphateles* (CDFW 2013), and the U.S. Fish and Wildlife Service (USFWS) intends to propose amending Part 17, Subchapter B of Chapter I, Title 50, of the Code of Federal Regulations to reflect the taxonomic change from *G. b. mohavensis* to *Siphateles bicolor mohavensis* (USFWS 2009). This taxonomic change will not affect its federal listing status. A physical description of the species can be found in the 5-Year Review (USFWS 2009).

The Mohave tui chub has a distinct lineage and is a separate subspecies from its closest relative, the Lahontan Lake and Lahontan

creek tui chubs (*Siphateles bicolor pectinifer* and *Siphateles bicolor obesa,* respectively). Mohave tui chub is least similar genetically to arroyo chub (*Gila orcutti*) (USFWS 2009).

Distribution

General

Historically, the Mohave tui chub is believed to have occurred throughout the Mojave River drainage (Miller 1946, cited in USFWS 1984). According to the Recovery Plan for the Mohave Tui Chub, *Gila bicolor mohavensis* (Recovery Plan) (USFWS 2009), the Mojave River drainage in the Mojave Desert originally consisted of the Mojave, Little Mojave, and Manix lakes; during the Pleistocene age, these lakes were connected through channels, and Mohave tui chubs were probably found throughout the drainage (Figure 2; USFWS 1984). As the climate became drier and the lakes receded, the Mohave tui chub was restricted to the Mojave River. During the 1930s, arroyo chubs were introduced into the Mojave River and likely hybridized with the Mohave tui chub, thus eliminating the genetically pure Mohave tui chub within the Mojave River (USFWS 1984). A small population of genetically pure Mohave tui chub persisted in isolated ponds near the terminus of the Mojave River at Soda Springs. Four populations of the Mohave tui chub have also been successfully introduced at the Lark Seep complex at China Lake Naval Weapons Station, Camp Cady Wildlife Area (USFWS 2009), the Lewis Center in Apple Valley, and Morning Star Mine at Mojave National Preserve. All of these populations are located within the Plan Area.

Distribution and Occurrences within the Plan Area

Historical

As described above, the Mohave tui chub was historically found within the Mojave River basin as the only native fish within this system. By 1970, the genetically pure Mohave tui chub had been eliminated from the Mojave River due to several factors, including hybridization; introduction of other non-native, competitive, and predatory aquatic species to its historical habitat (e.g., bass [*Micropterus* spp.], catfish [*Ictalurus* spp.], trout [*Oncorhynchus* spp.],

bullfrog [*Rana catesbeiana*], and crayfish [*Procambarus clarki*] [Miller 1969]); habitat alteration; water diversions; and pollution (USFWS 2009). At the time of listing in 1970, four populations were known to exist; three were located in San Bernardino County at Piute Creek, Two Hole Spring, and Soda Springs; and one was in Paradise Spa, Nevada (USFWS 2009). There are nine historical (i.e., pre-1990) records in the Plan Area contained in the California Natural Diversity Database, occurring in the eastern end of Mojave National Preserve and along the northern flank of the San Bernardino Mountains (Figure SP-F2) (CDFW 2013; Dudek 2013).

Recent

A population was established in 1978 at the Desert Research Station near Hinkley, California; however, in 1992 the pond dried up and the population was extirpated. As of 2011, there were five populations of genetically pure Mohave tui chubs: Soda Springs and Morning Star Mine at Mojave National Preserve, Lark Seep at China Lake Naval Air Weapons Station, Camp Cady Wildlife Area, and the Lewis Center in Apple Valley (Figure SP-F02). All of these locations are within the Plan Area. The Camp Cady Wildlife Area is managed by CDFW; Soda Springs Mojave National Preserve and Morning Star Mine are managed by the National Park Service; and the Lark Seep complex is located on a naval base managed by the Department of Defense.

Natural History

Habitat Requirements

Historically, within the Mojave River, the Mohave tui chub was associated with deep pools and sloughs of the river and was not found very far into small tributaries (USFWS 1984). Although the Mohave tui chub does not currently occupy the Mojave River, a few perennial stretches of the river remain that could support a fishery. The habitat requirements for this species include configuration, ecology, and water quality (Archbold 1996, cited in USFWS 2009). The configuration of a lacustrine pond or pool should include a minimum water depth of 4 feet with some freshwater flow for a mineralized and alkaline environment (USFWS 2009; NatureServe 2011). The pools or ponds should include some aquatic plants (e.g.,

Ruppia maritima, *Typha* spp., and *Juncus* spp.), which provide habitat for aquatic invertebrates consumed by Mohave tui chub and a substrate for egg attachment (USFWS 2009). Aquatic ditchgrass (*Ruppia maritima*) appears to be the preferred vegetation for egg attachment and thermal refuge in summer months (USFWS 1984). In addition, the Mohave tui chub is sensitive to predation from other fish species, and pools should be relatively free of arroyo chubs and other non-native aquatic wildlife species (USFWS 2009). Finally, to be suitable for Mohave tui chub, the water should have water quality parameters within the tolerable range for this species and be free of toxic substances or the threat of toxic substance spills (USFWS 2009). Water quality parameters include a temperature range from 37° Fahrenheit (F) to 97°F, dissolved oxygen at greater than 2 parts per million, a salinity of 40 to 323 milliosmols per liter, and a pH of up to 9 with 10 being tolerable for a short period of time (Feldmeth et al. 1985; Archbold 1996; and McClanahan et al. 1986, cited in USFWS 2009).

The current populations are located in primarily man-made or man-supported habitats. The population in Lark Seep is in a perennial body of water that is fed from the wastewater treatment facility in Ridgecrest, California. The population at Camp Cady is located in a man-made, lined pond that receives water from a pump. The populations at Soda Springs occur in two bodies of water, one is a man-made pond that receives water from a pump, and the other is an isolated spring on the edge of Soda Lake (USFWS 2009). The population at the Lewis Center is in two small man-made ponds with water supplied from a pump, and at Morning Star Mine, the population is in a man-made pond created by a perched aquifer. Table 1 lists primary habitat associations and parameters for Mohave tui chub.

Table 1. Habitat Associations for Mohave Tui Chub

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Lacustrine ponds/pools	All life history phases	Primary	Minimum depth of 4 feet and water quality limitations	USFWS 1984, 2009

Foraging Requirements

Not much is known about the specific diet of the Mohave tui chub. They forage on a variety of aquatic invertebrates, including plankton and insect larvae, small fish and organic detritus (Archdeacon 2007, cited in USFWS 2009; NatureServe 2011). Ponds and pools that have aquatic vegetation provide habitat for these food sources, as discussed previously under Habitat Requirements (USFWS 2009).

Reproduction

Mohave tui chubs spawn after 1 year of age (USFWS 1984). Spawning begins during the spring in March and April when water temperatures are warm enough (64° F) (Vickers 1973, cited in USFWS 1984). Spawning may occur in the fall as well. Egg masses are laid in vegetation where they become attached after fertilization. The eggs are approximately 0.04 inch in diameter and hatch after approximately 6 to 8 days when water temperatures are between 64° F and 68° F (USFWS 1984).

Spatial Activity

Currently, the populations of Mohave tui chub are restricted to ponds and man-made channels where they do not have any connection to other populations. Past efforts to introduce or transplant additional populations generally have not been successful (USFWS 2009) with the exception of their current locations in Kern and San Bernardino Counties, California.

Ecological Relationships

The Mohave tui chub originated from the Mojave River basin where it was adapted to the perennial deep pools and slough-like areas of the Mojave River and an absence of aquatic predators. Several factors contributed to its decline and current status as a federal and state listed species. The introduction of arroyo chub into the Mojave River in the 1930s resulted in likely hybridization and elimination of genetically pure Mohave tui chub species. The arroyo chub was also a source of competition for food.

Flooding, changes in water quality, and the introduction of non-native plant and wildlife species have also affected this species (USFWS 1984). Flooding in the Mojave River in 1938 enabled arroyo chubs to disperse further throughout the Mojave River system, and because of their adaptation to waters with greater velocities, the arroyo chub was successful at surviving these floods. Mohave tui chubs, on the other hand, are adapted to lacustrine conditions and are not able to persist in conditions with high-velocity flow and warmer shallow channels (USFWS 2009). These adaptive differences have contributed to replacement of Mohave tui chub by arroyo chub (Castleberry and Cech 1986). In addition, changes in water quality and quantity have resulted in the loss of subpopulations at East Pond (Camp Cady) and Three Bats Pond (Soda Springs) (USFWS 2009). The introduction of non-native plants and aquatic and amphibious species into the Mojave River system has resulted in modification of the species' habitat. Predation by introduced aquatic species (e.g., bass [*Micropterus* spp.], trout [*Oncorhynchus* spp.], catfish [*Ictalurus* spp.], mosquitofish [*Gambusia affinis*], and bullfrogs [*Lithobates catesbeianus*]) contributed to the extirpation of the Mohave tui chub in the Mojave River (USFWS 2009). The establishment of salt cedar (*Tamarix* sp.), has altered water flow and geomorphology of the Mojave River system (Lovich 2006).

A study conducted at Fort Soda in 1981–1982 found that Mohave tui chub populations increased two to three times during the spring and summer months, and then decreased during the fall and winter months (Taylor 1982). A study examining the growth and population structure of the Mohave tui chub at a research station northwest of Barstow in the 1980s found that the population was highest in late summer and lowest in late winter (Havelka et al. 1982). Tui chubs gained weight in May, but lost up to 35% of their body weight from June to October before gaining weight again in November. This may be the result of higher metabolic rates during the summer coupled with a possible reduction in planktonic biomass (Havelka et al. 1982).

Population Status and Trends

Global: Critically imperiled (NatureServe 2011)
State: Same as above
Within Plan Area: Same as above

As described previously under Distribution, Mohave tui chub is only present at five locations, and remains extirpated from its historic habitat in the Mojave River. As concluded in the 2009 5-Year Review for the species, the Mohave tui chub "still meets the definition of endangered in the Act for the following reasons: (1) there are fewer populations of this subspecies now than at the time of listing; (2) the rare nature of this subspecies increases the risk of local extirpations from stochastic events; (3) all populations of the Mohave tui chub are threatened by one or more of the threats described in the Recovery Plan that contributed to its endangered status including habitat loss and alteration, predation from non-native species, with the additional, newly identified threats of parasitism, genetic drift, and extirpation from stochastic events; (4) the lack of consistent and reliable management and monitoring activities for these populations, which makes it difficult to identify and determine the magnitude and imminence of current threats, and therefore, to ensure that the threats will be identified in time and ameliorated; and (5) the failure to meet any of the downlisting or delisting criteria in the Recovery Plan" (USFWS 2009).

Threats and Environmental Stressors

The American Fisheries Society publication of its endangered, threatened, or of special concern fishes of North America identified two main threats to Mohave tui chub: 1) the present threatened destruction, modification, or curtailment of its habitat or range; and 2) other natural or man-made factors affecting its continued existence (hybridization, introduction of non-native or transplanted species, predation, or competition) (Williams et al. 1989, cited in USFWS 2009).

The Mohave tui chub is already extirpated from its historical distribution in the Mojave River. As one of the criteria for delisting the Mohave tui chub, the Recovery Plan includes the return of the Mohave tui chub into its historical range in the Mojave River. Over the years,

the aquifer of the Mojave River has been overdrafted, resulting in the loss of aquatic habitat. Many of the areas within the river are now shallow and lack the lacustrine conditions once characteristic of portions of the Mojave River drainage, thus reducing the suitable habitat available for Mohave tui chub reintroduction.

A parasitic Asian tapeworm was found in Lake Tuendae (Soda Springs), and it initially had a deleterious effect on the population there. It was found to contribute to a reduced growth rate of Mohave tui chub in captivity, but not the survival rate (Archdeacon 2007). Research on Asian tapeworm parasitism has shown no long-term debilitating impacts on Mohave tui chub populations (Archdeacon 2007, cited in USFWS 2009).

Non-native species, such as bullfrogs and sport fish (e.g., bass and catfish), were introduced into the river. Predation on Mohave tui chub from these species contributed to its extirpation within the Mojave River (Williams et al. 1989, cited in USFWS 2009). Mosquitofish were found in Lake Tuendae (Soda Springs) in 2001 and were found to reduce the survival rate of the chubs when no cover is provided in the environment (Archdeacon 2007). They also compete for food and other resources, which may pose a threat to the Mohave tui chub.

Other threats to the Mohave tui chub include regulatory mechanisms. For example, USFWS (2009) states that the military installations do not obtain incidental take permits under the California Endangered Species Act; however, China Lake Naval Air Weapons Station implements Section 7(a)(1) of the federal Endangered Species Act, which requires federal agencies to utilize their authorities in the furtherance of the purposes of the act by carrying out programs for the conservation of federally endangered and threatened species. It should be noted that at the time of the 5-Year Review, the only proposed activities that would result in the take of Mohave tui chub were for research permits, which is purposeful take (USFWS 2009).

Conservation and Management Activities

The USFWS and cooperating agencies have proposed establishing additional populations of Mohave tui chub in the Mojave River watershed and the California portion of the Mojave Desert in order

to contribute to the conservation of the Mohave tui chub (USFWS 2011). An environmental assessment has been completed to analyze the locations where these populations could be established (USFWS 2011).

Because all of the current populations of Mohave tui chub occur in man-made or man-supported environments, ongoing conservation and management activities are required. To ensure the long-term sustainability of the Mohave tui chub, the 5-Year Review indicates that habitat management, ecosystem restoration, monitoring, and adaptive management are needed (USFWS 2009).

All of the current populations require regular control of cattails (*Typha* spp.) in ponds to maintain open water environments and suitable water conditions. Other specific management considerations include the Asian tapeworm, mosquitofish, habitat loss and degradation, water quality and supply, and genetic drift (USFWS 2009). Genetic drift can result in a loss of alleles (i.e., genetic variation) at small, isolated populations and can result in increased risk of extirpation. Recent data indicate that populations at MC Spring (at Soda Springs) and Camp Cady have recently shown a loss of genetic diversity (S. Parmenter, pers. comm. 2007, cited in USFWS 2009).

Data Characterization

To better manage and recover the species, the 5-Year Review (USFWS 2009) suggests identifying the extent and magnitude of bird predation, determining spawning requirements and early life history, determining physiological tolerances of Mohave tui chubs and arroyo chubs to water quality parameters, and identifying genetic issues, such as founder effect and possible hybridization with arroyo chubs.

Recent genetic analysis indicates that all existing populations of Mohave tui chubs are genetically pure; they don't show genetic evidence of hybridization with arroyo chubs. While the Mohave tui chub populations at Lark Seep and the Lake Tuendae subpopulation of Soda Springs are heterogeneous, genetic drift, or a loss of alleles, has occurred at the MC Spring subpopulation of Soda Springs and Camp Cady (USFWS 2009).

Hybridization between Mohave tui chub and the Los Angeles Basin endemic arroyo chub was identified as a primary threat to the Mohave tui chub after arroyo chubs were introduced to the Mojave River in the 1930s. However, hybridization between these two fish has never been studied and documented. Mojave National Preserve has initiated research on the ability of these two fish to hybridize (USFWS 2009).

Management and Monitoring Considerations

Management and monitoring considerations are addressed in the Recovery Plan (USFWS 1984) and 5-Year Review (USFWS 2009) as actions necessary to downlist and delist the species. The overall objective of the Recovery Plan for delisting is to reintroduce a viable, sustainable population of Mohave tui chub into a majority of its historic habitat in the Mojave River (USFWS 1984). To achieve this objective, several management activities must occur, including management of introduced aquatic predators, hybridization with arroyo chub, water supply, water quality, and suitable habitat (e.g., deep, cool pools and sloughs).

In the interim, the Recovery Plan identified objectives to downlist the species from endangered to threatened. These objectives include establishing six populations of at least 500 Mohave tui chub in each population. Currently, there are only three populations that meet this criterion. Portions of the Mojave River that have been identified for additional potential reintroduction include the Mojave Narrows Regional Park area in Victorville, Camp Cady, portions of Afton Canyon, and an area downstream from the Victor Valley wastewater treatment facility in Oro Grande (USFWS 2009). However, it is likely that habitat management of these areas would be required because many of them have shallow flows rather than the preferred habitats of pools and sloughs.

Because all of these areas identified for potential reintroduction are located within the Plan Area, there should be careful consideration of future activities that could affect these areas.

Furthermore, the American Fisheries Society has published guidelines for introductions of threatened and endangered species that could be applied to Mohave tui chub (Williams et al. 2011). The guidelines

recommend restricting introductions to sites within the native or historic habitat, sites that are protected, sites where the potential for dispersal has been determined acceptable, sites that fulfill the species' life history requirements, and sites that contain sufficient habitat to support a viable population. In addition, introduction sites should be avoided where endangered or threatened fish could hybridize with other taxa or where other rare or endemic taxa could be adversely affected. The introduction stock should be from an appropriate source, should be examined for taxonomic status and presence of undesirable pathogens, should be of sufficient number and character, should be carefully and quickly transported, should be introduced under favorable conditions, and the translocation procedures should be documented. After translocation, the American Fisheries Society recommends systematic monitoring of introduced populations, which involves restocking if necessary, determining the cause of any failures, and documenting findings and conclusions reached during the post-introduction (Williams et al. 2011).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Mohave tui chub, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 360 acres of modeled suitable habitat for Mohave tui chub in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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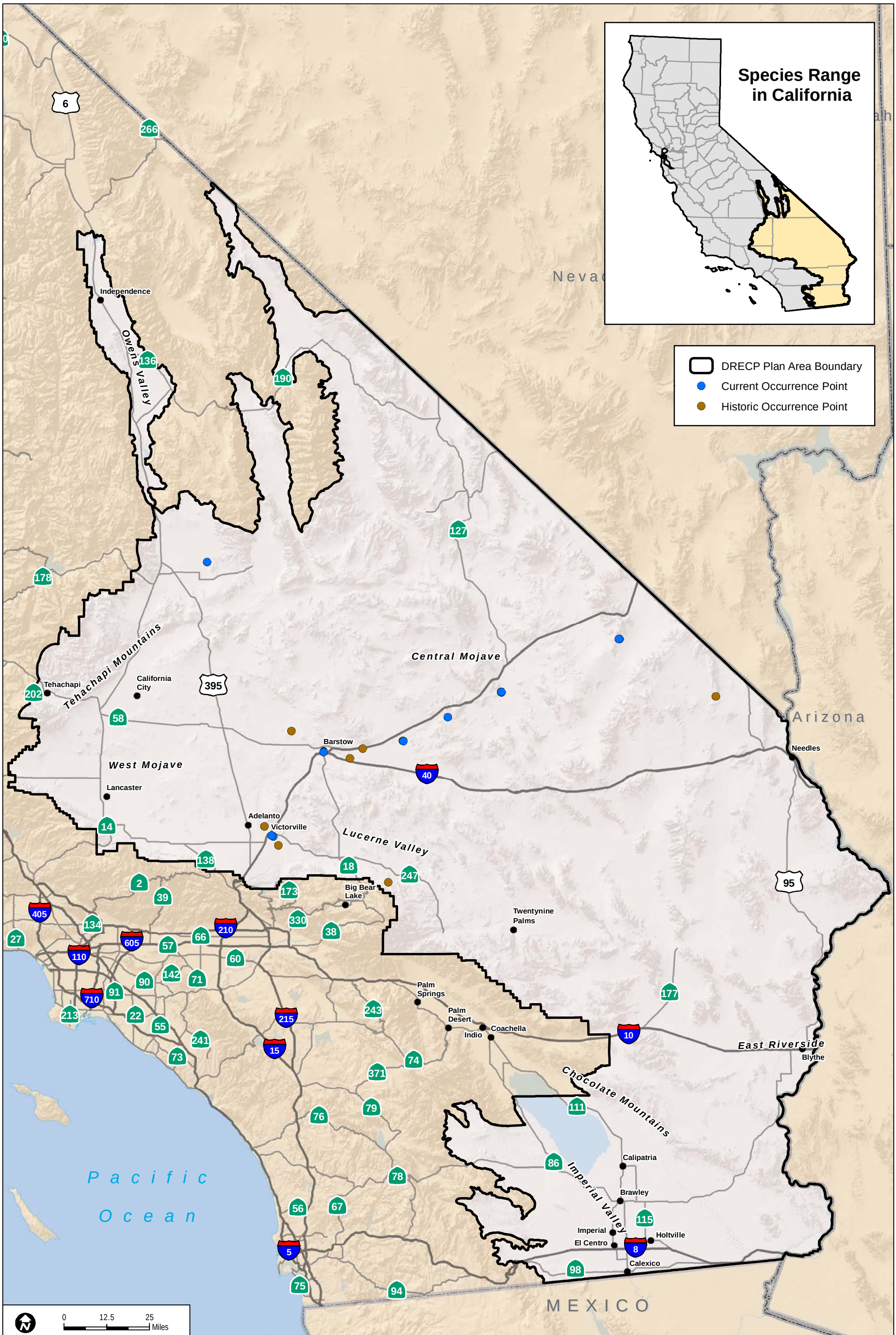

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-F02

Mohave Tui Chub Occurrences in the Plan Area

Owens Pupfish (*Cyprinodon radiosus*)

Legal Status

State: Endangered, Fully Protected
Federal: Endangered
Critical Habitat: N/A
Recovery Planning: *Owens Basin Wetland and Aquatic Species Recovery Plan, Inyo and Mono Counties, California* (USFWS 1998)
Notes: Species was federally listed endangered on March 11, 1967. It was listed as endangered in California in 1971 (USFWS 2009).

Taxonomy

The first taxonomic description of Owens pupfish (*Cyprinodon radiosus*) was in 1948 by Miller, but occurrence locations along with relative abundance observations of Owens pupfish were noted as early as 1859 by explorers and scientists (USFWS 2009). Owens pupfish is in the killifish family (Cyprinodontidae) and is one of five pupfish species native to California (BLM 2011). The desert pupfish (*Cyprinodon macularius*), which occurs in the lower Colorado River system, is the closest relative of the Owens pupfish (USFWS 1998). Though Owens pupfish is a member of the *C. nevadensis* complex, a group of four species in two closed basins of the Death Valley System in California and Nevada (Owens River Valley and Ash Meadows–Death Valley), it appears to be more closely related to *C. macularius* than to the Ash Meadows–Death Valley members of the complex (*C. diabolis, C. nevadensis,* and *C. salinus*). Apparently, *C. radiosus* and *C. macularius* share both a general morphological similarity and an ancestral mitochondrial deoxyribonucleic acid (DNA) that separates them from the Ash Meadows–Death Valley pupfishes (Echelle and Dowling 1992). Descriptions of the species' physical characteristics can be found in the U.S. Fish and Wildlife Service (USFWS) 5-Year Review (2009).

Distribution

General

The Owens pupfish is restricted to the Owens Valley portion of the Owens River in Mono and Inyo counties, California (Figure SP-F3). Based on historical observations, Owens pupfish is believed to have occupied all of the Owens River and possibly the Owens River Delta at Owens Lake. Currently, it occurs at Fish Slough, Mule Springs, Well 368, and Warm Springs (USFWS 2009). Eight of the 17 California Natural Diversity Database (CNDDB) occurrences are within the Plan Area, while the remaining occurrences are farther north and east of the Plan Area (CDFW 2013).

Distribution and Occurrences within the Plan Area

Historical

Five of the eight occurrences in the Plan Area were last documented prior to 1990 (Figure SP-F03). All of these are found within the Owens Valley in Inyo County and have possibly been extirpated (CDFW 2013; Dudek 2013).

Recent

Three recent occurrences (i.e., since 1990) of Owens pupfish occur in the Plan Area. One occurrence is at Well 368, located 0.2 mile west of the Owens River and 2.5 miles south of Mazourka Canyon Road. Last observed in 1999, this occurrence is presumed extant. In 1988, pupfish from Warm Springs were introduced into the ponds at this location, and both adults and juveniles were abundant throughout the North Fork Area in 1999. It is owned by the Los Angeles Department of Water and Power (CDFW 2013; Dudek 2013).

Natural History

Habitat Requirements

Owens pupfish occurs in shallow water habitats in the Owens Valley (CDFW 2013). It will occupy most aquatic habitat where water is

relatively warm and food is plentiful (USFWS 2009). However, it prefers warm, clear, shallow water, free of exotic fishes, and requires areas of soft substrate for spawning (CDFW 2013; USFWS 2009). In addition, Owens pupfish habitat differs from the habitat of other pupfish. Specifically, aquatic habitats associated with the Owens River are typically colder, frequently covered by ice during winter, and lower in conductivity and salinity than habitats occupied by other pupfish species (USFWS 2009).

All life stages may be found in the various microhabitats available with little apparent documented preference. However, adults frequently occupy deeper water than juveniles. Male pupfish are territorial and defend areas of substrate from competing males. Females occupy habitats along the margins of these territories (USFWS 2009). Table 1 lists primary habitat associations and parameters for Owens pupfish.

Table 1. Habitat Associations for Owens Pupfish

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Warm, clear, shallow aquatic habitat	Primary	Soft substrates required for spawning	CDFW 2013; USFWS 2009

Notes: Species only occurs in the Owens River.

Foraging Requirements

Owens pupfish are opportunistic omnivores and consume a variety of plant and animal foods. Their diet changes seasonally and generally includes whatever invertebrates and plants are most abundant at that time (USFWS 1998). However, they primarily feed on aquatic insects and are an effective biological control agent for mosquitos (USFWS 2009; USFWS 1998). They do not prey on other fishes (USFWS 1998).

Reproduction

Owens pupfish breed from April through October (BLM 2011). Females spawn over soft substrates in spring and summer when water temperatures are near 14°Celsius (C) (57°Fahrenheit [F])

(USFWS 1998). They may spawn up to 200 times per day, laying one or two eggs at a time (USFWS 2009). Males are very aggressive during the breeding season as they protect their breeding territory (BLM 2011). Incubation lasts for approximately 6 days before hatching in water that ranges in temperature from 75°F to 81°F. On average, 95% of spawned eggs are fertilized. Juvenile pupfish reach sexual maturity in 3 to 4 months and are generally able to spawn before their first winter (USFWS 2009).

In a study examining Owen's pupfish mating systems and sexual selection, it was found that the size of the mother did not strongly influence egg size or fry size. In addition, individual egg size was not correlated with fry size (Mire and Millett 1994).

Spatial Activity

Little information is known regarding this species' spatial activity. However, CDFW (2013) refers to migration between areas. As noted previously in Habitat Requirements, males are territorial and females occupy areas at the margins of territories.

Ecological Relationships

Generally, the lifespan of Owens pupfish is rarely over 1 year. However, they live up to 3 years in refuge habitats (USFWS 2009).

Owens pupfish congregate in small schools (USFWS 2009). Owens pupfish demography has been studied only in intensively managed refuge habitats with little environmental variation. Demographic studies of other pupfishes in the Death Valley system, however, suggest large seasonal variation in population size. Although studies of Owens pupfish in managed refuge habitats indicate little seasonal variation in population size, unmanaged populations may experience more temporal variation in habitats that are more representative of areas historically occupied (USFWS 2009).

Owens pupfish scarcity in the 1930s was attributed to establishment of non-native predatory fish. In addition, water diversions that decreased and altered Owens River flows desiccated shallow pupfish habitats bordering the river (USFWS 1998).

Population Status and Trends

Global: G1, Critically imperiled (NatureServe 2011, conservation status last updated 2007)
State: S1, Critically Imperiled (CDFW 2013)

By the 1930s Owens pupfish was scarce throughout most of its historical range. It was believed to be extinct from 1942, until in 1964 when a single population of approximately 200 fish was rediscovered in Fish Slough (USFWS 1998). This was the only known existing population when Owens pupfish was listed as federally endangered in 1967. This population still persists today (USFWS 2009). Since its listing, three additional populations have been established at Warm Springs, Well 368, and Mule Springs, (USFWS 2009). These additional existing populations were established from progeny of the remnant population at Fish Slough (USFWS 1998). All existing populations are small, ranging from 100 to 10,000 individuals. The Owens pupfish still faces a high degree of threat, but it also has a high recovery potential (USFWS 2009).

Threats and Environmental Stressors

The 1998 Recovery Plan states that Owens pupfish is affected by non-native species and habitat modification for water diversions that altered Owens River flows (USFWS 1998, 2009). Currently, all populations of Owens pupfish are threatened by loss of habitat resulting from cattail (*Typha* spp.) encroachment. Emergent vegetation and accumulated detritus covers and reduces the substrate used by the pupfish for breeding. Emergent vegetation also reduces water depth, elevates water temperature, and potentially produces severe anoxic conditions (USFWS 2009).

Owens pupfish is also seriously threatened by non-native predators. Because populations are highly localized and relatively small, they can be threatened by a single individual predator. At the time of listing in 1967, several non-native fish predators affecting Owens pupfish were identified: largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieui*), brown trout (*Salmo trutta*), and bluegill (*Lepomis macrochirus*). Since its listing, mosquitofish (*Gambusia affinis*), crayfish (*Pastifasticus leniusculus*), and bullfrogs (*Rana*

catesbeiana) have been introduced into the pupfish's habitat and also threaten Owens pupfish. Besides eating young and adult Owens pupfish, non-native predators compete with Owens pupfish for food and habitat (USFWS 2009).

Additionally, the Owens pupfish is highly vulnerable to extinction from stochastic (random) demographic, genetic, and catastrophic environmental events because the existing populations are small and isolated. Demographic stochasticity refers to random variability in survival and/or reproduction among individuals that can have a significant impact on population viability when populations are small and short-lived with low fecundity (reproductive output). Genetic stochasticity results from the changes in gene frequencies caused by the loss of genetic variation when a new population is established by a very small number of individuals (i.e., the founder effect). This can result in random gene fixation in which some portion of gene loci are fixed at a selectively unfavorable allele (a different form of a gene) because natural selection is not intense enough to overcome random genetic drift. Inbreeding bottlenecks in which a significant percentage of a population is killed or prevented from breeding may also occur in small, isolated populations. Environmental stochasticity is the variation in birth and death rates from one season to the next in response to weather, disease, competition, predation, or other external factors. These three factors may act alone or in combination to reduce the long-term viability of small populations (USFWS 2009).

Conservation and Management Activities

Owen's pupfish reestablishment in the Owens Valley Native Fish Sanctuary has developed as a cooperative undertaking between the City of Los Angeles and the California Department of Fish and Wildlife (CDFW) (Miller and Pister 1971). USFWS and CDFW are making progress toward establishing two new pupfish populations. These populations will be established at the Cartago Springs Wildlife Area (USFWS 2009). Although the four existing Owens pupfish populations do not have approved management plans or implementing agreements between the USFWS and landowners, the new pupfish populations would require management plans that would address threats (USFWS 2009).

Fish screens and the isolation of the artificial refuges for Owens pupfish populations provided some protection from non-native fish predators. In addition, the CDFW actively removes predators as they are observed. Despite these efforts, predators are likely reintroduced into Owens pupfish populations by fishermen intending to stock those sites with bait and sport fish. Cattail encroachment is currently managed at all populations. If not actively managed, cattails will grow back and threaten Owens pupfish breeding sites (USFWS 2009).

Data Characterization

Few studies have examined the ecology of Owens pupfish. Owens pupfish demography has been studied only in intensively managed refuge habitats that may not be representative of the species' historical, natural demography (USFWS 1998).

Management and Monitoring Considerations

Habitat protection and management is the key to the recovery of the Owens pupfish (BLM 2011). The 5-Year Review for the species includes the following recommendations for actions over the next 5 years (USFWS 2009):

1. Remove emergent vegetation and eradicate non-native predators from Warm Springs and reestablish Owens pupfish in the upper and lower ponds
2. Evaluate Round Valley to determine if it is a suitable location for a population of Owens pupfish
3. Develop management plans and implementation agreements for all populations
4. Establish a new population of Owens pupfish at Cartago Springs Wildlife Area and Blackrock Waterfowl Management Area
5. Conduct population surveys and demographic studies, collect additional genetic samples, and complete genetic analysis. Develop breeding programs based on the results of genetic analysis to optimize genetic material in all populations of Owens pupfish.

Furthermore, the American Fisheries Society has published guidelines for introductions of threatened and endangered species that could be applied to Owens pupfish (Williams et al. 2011). They recommend restricting introductions to sites within the native or historic habitat, sites that are protected, sites where the potential for dispersal has been determined acceptable, sites that fulfill the species' life history requirements, and sites that contain sufficient habitat to support a viable population. In addition, introduction sites should be avoided where endangered or threatened fish could hybridize with other taxa or where other rare or endemic taxa could be adversely affected. The introduction stock should be from an appropriate source, should be examined for taxonomic status and presence of undesirable pathogens, should be of sufficient number and character, should be carefully and quickly transported, should be introduced under favorable conditions, and the translocation procedures should be documented. After translocation, the American Fisheries Society recommends systematic monitoring of introduced populations, which involves restocking if necessary, determining the cause of any failures, and documenting findings and conclusions reached during the post-introduction (Williams et al. 2011).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Owens pupfish, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 17,547 acres of modeled suitable habitat for Owens pupfish in the Plan Area. Appendix C includes specific model parameters and a figure showing the modeled suitable habitat in the Plan Area.

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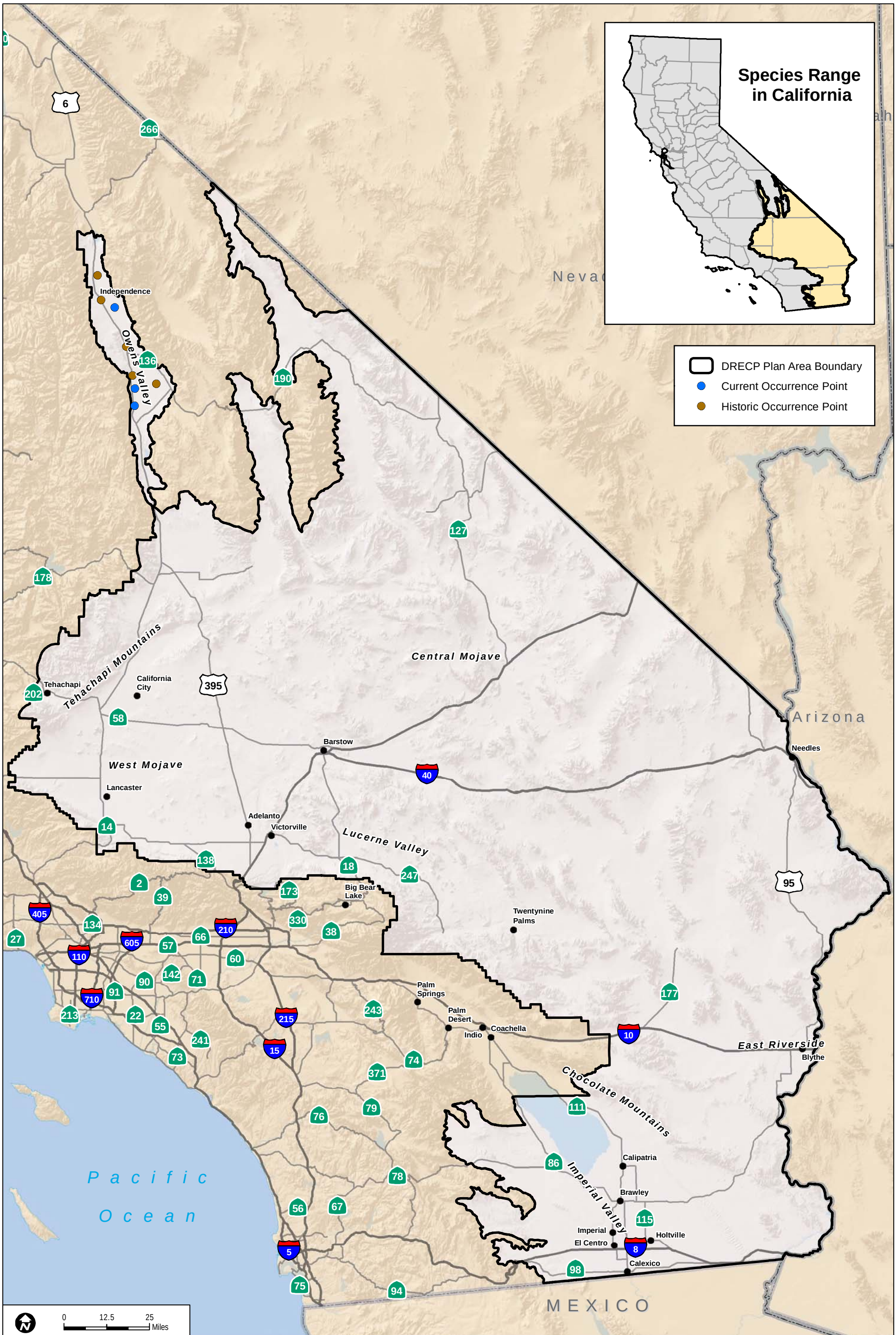

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-F03

Owen's Pupfish Occurrences in the Plan Area

Owens Tui Chub (*Siphateles bicolor snyderi = Gila bicolor snyderi*)

Photo courtesy of Joe Ferreira

Legal Status

State: Endangered,
Fully Protected
Federal: Endangered
Critical Habitat: Designated on August 5, 1985 (50 FR 31592–31597)
Recovery Planning: Owens Basin Wetland and Aquatic Species Recovery Plan, Inyo and Mono Counties (USFWS 1998)
Notes: The 5-year review for this species (USFWS 2009) found that threats that were present when the Owens tui chub was listed are still present with new threats identified. The recovery priority number assigned was 3, which indicates the taxon is a subspecies that faces a high degree of threat and has a high potential for recovery (USFWS 2009).

Taxonomy

The Owens tui chub (*Siphateles bicolor snyderi*) is a member of the minnow family (*Cyprinidae*). It was described in 1973 as a subspecies of tui chub endemic to the Owens Basin (Miller 1973) as *Gila bicolor snyderi*. Simons and Mayden (1998) published a paper addressing the classification of the North America genera of *Cyprinidae* and, based on ribonucleic acid sequences, restored *Siphateles* from a subgenus to a full genus. The California Department of Fish and Game (CDFG) currently includes the species under the genus *Siphateles* (CDFG 2011), and the U.S. Fish and Wildlife Service (USFWS) proposes the scientific name change from *G. b. snyderi* to *S. b. snyderi* (USFWS 2009). This name change will not affect its federal listing status.

It is morphologically similar to the Mohave tui chub (*S. b. mohavensis*) and Lahontan tui chub (*S. b. obesus*). It is distinguished from its closest relative, the Lahontan tui chub, by scales with a weakly developed or absent basal shield, lateral and apical radii that number 13 to 29, the

structure of its pharyngeal arches, the number of anal fin rays, gill-raker counts of 10 to 14, and 52 to 58 lateral line scales (Miller 1973). Dorsal and lateral coloration varies from bronze to dusky green, grading to silver or white on the belly. The species may reach a total length of 12 inches. The Owens tui chub evolved in the Owens River watershed with only three other smaller species of fishes, Owens pupfish (*Cyprinodon radiosus*), Owens speckled dace (*Rhinichthys osculus* ssp.), and Owens sucker (*Catostomus fumeiventris*) (USFWS 2009).

Based on recent genetic research, Chen et al. (2007) proposed that the Cabin Bar Ranch population is a separate lineage—the Toikona tui chub lineage—from the Owens tui chub lineage. They do not propose making a formal taxonomic split from the Owens tui chub until more information becomes available.

Descriptions of the species' physical characteristics can be found in USFWS (1998) and USFWS (2009).

Distribution

General

The Owens tui chub is endemic to the Owens Basin (Owens Valley, Round Valley, and Long Valley) of Inyo and Mono Counties, California (CDFW 2013; USFWS 1998).

Distribution and Occurrences within the Plan Area

Historical

Early fish collections in the Owens Basin documented Owens tui chub in Owens Lake, several sites along the Owens River from Long Valley to Lone Pine, tributary streams near the Owens River in Long Valley and Owens Valley, Fish Slough, and irrigation ditches and ponds near Bishop, Big Pine, and Lone Pine (Miller 1973; USFWS 2009). Although there are only two historical (i.e., pre-1990) records for Owens tui chub in the Plan Area in the California Natural Diversity Database (CNDDB) (Figure SP-F04) (CDFW 2013; Dudek 2013), the scattered distribution of these localities and the ease with which researchers captured fish suggest that

Owens tui chub were common and occupied all valley floor wetlands near the Owens River in Inyo and Mono counties (USFWS 2004).

Recent

Currently, genetically pure Owens tui chub is limited to six isolated sites in the Owens Basin: Hot Creek Headwaters (AB Spring and CD Spring), Little Hot Creek Pond, Upper Owens Gorge, Mule Spring, White Mountain Research Station (operated by the University of California), and Sotcher Lake, the last of which is outside the historical range of the species in Madera County (USFWS 2009). However, there are only three recent occurrence records documented in the CNDDB database (Figure SP-F04; CDFW 2013; Dudek 2013). In 1987, Owens tui chub were found occupying irrigation ditches and a spring at Cabin Bar Ranch on the southwest shore of Owens Dry Lake, and became known as the Cabin Bar Ranch population (USFWS 2009). Predation from introduced largemouth bass (*Micropterus salmoides*) and bluegill sunfish (*Lepomis macrochirus*), and failure to maintain adequate water quality and quantity, extirpated the Cabin Bar Ranch population of Owens tui chub in 2003 (USFWS 2009). However, prior to extirpation, 24 individuals were placed in an artificial pond and moved to Mule Spring in 1990; all extant fish of this group descend from this transplant (Chen et al. 2007). The Plan Area only includes the former Cabin Bar Ranch population, with the Mule Spring population (see Figure SP-F04) adjacent and outside of the Plan Area boundary. USFWS (1998) has proposed two conservation areas within the Plan Area: Black Rock and Southern Owens Dry Lake (the Cabin Bar Ranch population was found on the southwest shore of Owens Dry Lake).

Natural History

Habitat Requirements

The Owens tui chub occurs in low-velocity waters with well-developed beds of aquatic plants, rocks, and undercut banks with bottoms of gravel (Leunda et al. 2005; Moyle 2002). Dense aquatic vegetative cover is likely important to Owens tui chubs for predator avoidance, reproduction, water velocity displacement, and feeding (McEwan 1989, as cited in Geologica 2003; McEwan 1991). Plant

species observed in occupied habitat at the Hot Creek Headwaters population include watercress (*Nasturtium officinale*), water fern (*Azolla filiculoides*), duckweed (*Lemna* sp.), pondweed (*Potamogeton* sp.), aquatic buttercup (*Ranunculus aquatilis*), and elodea (*Elodea canadensis*) (McEwan 1991). McEwan (1991) provides details of the habitat structure at the Hot Creek Headwaters population, where plants cover approximately 50% to 75% of the stream surface area. The plants typically grow out from the sides in the main channel, forming dense beds along the stream margins that delineate a small chute of swift-flowing water in the center of the channel. In the backwater areas with zero water velocities, vegetation covers nearly 100% of the surface area. There is a limited die-off of vegetation beds during the winter, but most of the beds persist due to the thermal characteristics of the headsprings.

Water temperature within occupied habitat varies to a great degree (as summarized in Geologica [2003]). It can be fairly constant at spring sites (14–18°C [57–64°F]), hotter at hot springs (21–25 °C [70–77°F]), and cooler in a river (36–78°F [2–25°C]) (Geologica 2003). Within occupied habitat where measurements exist, pH ranges from 6.6 to 8.9 (McEwan 1989; Geologica 2003), dissolved oxygen varies from 5 to 9.3 milligrams/liter (Malengo 1999; Geologica 2003), and alkalinity varies from 68.0 to 88.4 parts per million (McEwan 1989).

The Owens tui chub is restricted to six total populations, five of which are within the historical range of the species. Of these five populations, three (Hot Creek Headwaters, Little Hot Creek Pond, and Upper Owens Gorge) are located in small, isolated, man-altered portions of these waterways. The other two populations (Mule Spring and White Mountain Research Station) exist in manmade ponds at upland sites with water supplied by artificial methods. A detailed account of the habitat at each of the extant populations can be found in the 5-year review (USFWS 2009).

Table 1. Habitat Associations for Owens Tui Chub

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Low-velocity waters	Breeding /foraging	Primary	Low-velocity waters with well-developed beds of aquatic vegetation, rocks, and undercut banks	Direct observation studies

Sources: USFWS 2009; Leunda et al. 2005; McEwan 1991, Geologica 2003.

Foraging Requirements

The results of a gut content analysis indicate that Owens tui chub is an opportunistic omnivore that utilizes a wide variety of food items (McEwan 1991). Aquatic vegetation is especially important as it provides forage and habitat for aquatic invertebrates, the main food item of the Owens tui chub (McEwan 1989, as cited in Geologica 2003; McEwan 1991). Specific food items that appear to be of importance include chironomids, larvae of two species of hydroptillid caddisfly, other aquatic invertebrates, plant material, and detritus (McEwan 1991). There is evidence that the diet varies seasonally at the Hot Creek Headwaters (McEwan 1991); the dominant items in Owens tui chub diet there are chironomid larvae and algae in spring, chironomid larvae in summer, hydroptillid caddisflies in fall, and chironomid larvae in winter (McEwan 1991). Owens tui chubs feed mainly by gleaning and grazing among submerged vegetation (Geologica 2003).

Reproduction

Sexual maturity in Owens tui chub appears dependent on the microhabitat. For example, sexual maturity in springs with constant water temperature has been recorded at 2 years for females and 1 year for males, in comparison to more varied temperatures where males and females reach sexual maturity at 2 years (McEwan 1990, as cited in USFWS 2009). In general, tui chubs congregate from later winter to early summer to spawn over aquatic vegetation or gravel substrates (Kimsey 1954, as cited in Geologica 2003). More

specifically, McEwan (1990, as cited in USFWS 2009), recorded spawning from late winter to early summer at spring habitats, and from spring to early summer in riverine and lacustrine or lake-like habitats. Spawning appears to be triggered by day length and warming water temperatures (McEwan 1989, 1990, as cited in USFWS 2009). With the adhesive quality of the eggs, spawning usually occurs over gravel substrate or aquatic vegetation (USFWS 2009). Multiple spawning bouts during the breeding season are likely (Moyle 2002), and females may produce large numbers of eggs at each bout (Geologica 2003). Embryos hatch in 3 to 6 days (Moyle 2002), and may be influenced by water temperature, with eggs hatching earlier in warmer water (Cooper 1978, as cited in USFWS 2009). Larvae remain near aquatic plants after hatching (Moyle 2002). Growth during the first summer is rapid and slows at maturity, usually in the second to fourth year (Moyle 2002).

Table 2. Key Seasonal Periods for Owens Tui Chub

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding			X	X	X	X	X					

Sources: USFWS 1998, 2009.

Spatial Activity

The dispersal, home range, and migratory patterns of Owens tui chub are not well understood. Many of the locations where they are currently found are completely isolated from other populations. Tui chubs congregate from late winter to early summer to spawn over aquatic vegetation or gravel substrates (USFWS 2009). Chen et al. (2007) have determined that the Owens tui chub lineage is more genetically distinct from the Cabin Bar Ranch population (the Toikona tui chub lineage) than the Lahontan tui chub, which may represent independent lines of evolution (i.e., no dispersal). Morphology, swimming ability, and behavior all suggest the species is not adapted to movement through rapid waters (Moyle 2002). Therefore, movement of this species likely requires the presence of vegetation beds so that high-velocity areas are encountered only briefly. Jenkins (1990, as cited in Geologica 2003) observed no Owens tui chub in the Owens River

Gorge within riffle habitat. Dispersal of other species of tui chub has been inferred using gene flow, where unidirectional dispersal and bidirectional inter-basin gene flow have been recorded (Chen 2006). In addition, daily migrations have been observed for tui chub in large, deep lakes during summer, whereas they move between deep water during the day and shallow water during the night (Moyle 2002).

Ecological Relationships

Owens tui chub were once common and occupied all valley floor wetlands near the Owens River in Inyo and Mono counties. Since that time, predaceous non-native fishes, extensive development of water resources, and interbreeding with Lahontan tui chub has resulted in population decline and habitat loss.

Currently, the major threat to the species is introgression with Lahontan tui chub (Chen et al. 2007). The Owens tui chub is reliant on slow-moving freshwater habitats that provide food and cover, but that are free of non-native aquatic predators and other tui chub subspecies and hybrids. It requires aquatic vegetation for cover, foraging, and spawning, as well as gravel substrates for spawning. If one or more of these elements are absent, it can be quickly extirpated from a location.

Population Status and Trends

Global: Critically imperiled (NatureServe 2011)
State: Same as above
Within Plan Area: Same as above

Since its listing in 1985, three new populations of Owens tui chub have been established, bringing the current number to six. Four of these populations are in small, manmade or man-altered waters, and one is outside the historical range of the species at an artificial lake (Sotcher Lake). USFWS (2009) recommends that a Recovery Priority Number of 3 be assigned to Owens tui chub, which indicates that the taxon is a subspecies that faces a high degree of threat and has a high potential for recovery. The threats that were present when the Owens tui chub was listed are still present with new threats identified (USFWS 2009).

Threats and Environmental Stressors

USWFS (2009) provides a detailed explanation of the threats to Owens tui chub, which are summarized here. Currently, the major threat to the species is introgression (i.e., hybridization) with Lahontan tui chub (Chen et al. 2007), which has resulted in extirpation throughout most of its range (USFWS 2009). In 1973, the Lahontan tui chub was introduced as baitfish into many of the streams in the Owens Basin. Historically, the Owens tui chub and Lahontan tui chub were isolated from each other, but now hybridization has been documented for populations in Mono County—at Hot Creek (downstream from the hatchery), Mammoth Creek, Twin Lakes–Mammoth, June Lake, and Owens River Upper Gorge Tailbay. In Inyo County, hybridization has been documented at A1 Drain, C2 Ditch, and McNally Canal (Madoz et al. 2005, as cited in USFWS 2009; Chen 2006, as cited in USFWS 2009). If the barriers that are acting to isolate the Owens tui chub populations from Lahontan tui chub become permeable, this could result in the loss of genetically pure populations of Owens tui chubs at Hot Creek Headwaters, Little Hot Creek Pond, and the Upper Owens Gorge. In addition, the opportunities to establish new populations of Owens tui chub in the Owens Basin are limited by the presence of hybrids in the Owens River and its tributaries. Currently, the only viable locations for establishing the Owens tui chub are isolated springs or the headwaters of streams with downstream barriers to upstream movement of Lahontan tui chubs or hybrids.

USFWS (50 FR 31592–31597) identified extensive habitat destruction and modification as threats to the Owens tui chub, and this is current as of today. Currently, Owens Basin water is in high demand that is expected to increase, which would reduce the overall availability of surface waters. The survival of two populations (White Mountain Research Station and Mule Spring) is dependent upon the continual maintenance of the artificial water supply and assurance of adequate water quality. The Upper Owens Gorge population is a pool created by a beaver dam that is eroding, which is slowly reducing the lacustrine habitat for Owens tui chubs.

Submerged aquatic vegetation is a key habitat requirement for the Owens tui chub, but not with large amounts of emergent vegetation because it may provide cover for nonnative predators of Owens tui

chubs, such as bullfrogs and crayfish (*Procambarus* sp.). At the spring sites (Hot Creek Headwaters, Little Hot Creek Pond, and Mule Spring), emergent vegetation (e.g., cattail) have reduced and altered the aquatic habitat, and routine removal of emergent vegetation is required. The Mule Spring and White Mountain Research Station populations require routine management of water quantity and water quality. The environment that the Upper Owens Gorge population inhabits has been severely altered by the construction of a dam, with no mechanism to manage adequate releases of water downstream of the dam.

Since listing, evidence of disease has been observed in some populations of the Owens tui chub (USFWS 2009). In AB Spring at Hot Creek Headwaters, Bogan et al. (2002, as cited in USFWS 2009) found evidence of infection in six of the seven Owens tui chubs that were collected for genetic analysis. Since disease has been identified in Owens tui chubs, it is considered a threat. However, the magnitude of this threat is unknown (USFWS 2009).

The final listing rule (50 FR 31592–31597) identified predation by introduced non-native fish as a major threat to the Owens tui chub. Predation by non-native largemouth bass and brown trout is thought to have eliminated Owens tui chubs from much of their historical range in the Owens River (Chen and May 2003), and it is believed that non-native fish (largemouth bass and bluegill sunfish) played a role in extirpating the Cabin Bar Ranch population (Chen et al. 2007). Mosquito fish (*Gambusia affinis*) may also present a threat, as they are known to prey on small individuals of Mohave tui chub (Archdeacon 2007, as cited in USFWS 2009). At Mule Spring, bullfrogs are present and probably prey on Owens tui chubs, as they are known to prey on other subspecies of tui chubs (Parmenter 2006, as cited in USFWS 2009).

The inadequacy of existing regulatory mechanisms is considered a threat at this time by USFWS (2009), largely due to unregulated actions that could overdraft the aquifer in the Owens Valley Groundwater Basin area, which may result in reduced or no water flow to existing isolated springs and headwater springs of streams in the Owens Basin. The issue stems from the fact that the aquifer in the Owens Basin has not been adjudicated and its use is not regulated. Any reduction in flow from springs in the Owens Basin would result in

further reductions of habitat quality and quantity for the Owens tui chub at springs and tributaries of the Owens River.

Currently, Owens tui chub populations are small, between 100 and 10,000 individuals; therefore, random events that may cause high mortality or decreased reproduction could readily eliminate an entire population, which would have a significant effect on the viability of Owens tui chub populations. Furthermore, because the number of populations is small (six) and each is vulnerable to this threat, the risk of extinction is exacerbated (USFWS 2009). The Owens tui chub has experienced population loss from environmental stochastic events and will likely do so in the future. For example, the Cabin Bar Ranch population was lost because of an apparent failure to maintain adequate water quality and quantity and the introduction of non-native predators. Another example is the disappearance of Owens tui chub from the Owens Valley Native Fishes Sanctuary (Fish Slough). Reasons for the loss of this population are not known, but the small, isolated nature of this population likely contributed to their extirpation (USFWS 2009).

In small populations, such as the Owens tui chub, there are a number of factors that may reduce the amount of genetic diversity retained within populations and may increase the chance that deleterious recessive genes are expressed. Loss of diversity could limit the species' ability to adapt to future environmental changes and contributes to inbreeding depression (i.e., loss of reproductive fitness and vigor) (USFWS 2009). Deleterious recessive genes could reduce the viability and reproductive success of individuals. Isolation of the six remaining populations, preventing any natural genetic exchange, will lead to a decrease in genetic diversity.

Conservation and Management Activities

The recovery plan (USFWS 1998) provides a detailed account of management goals that need to be successfully implemented in order for the species to be delisted:

- Establish multiple, self-sustaining populations of Owens tui chubs throughout much of the historical range of the species in six identified conservation areas;

- Ensure these populations are self-sustaining;
- Ensure that each population contains juvenile and three additional age classes, and that the biomass of Owens tui chubs exceed the biomass of deleterious, non-native aquatic predatory species, which would demonstrate successful recruitment and minimal predation on smaller Owens tui chubs by non-native aquatic species;
- Reduce competition with non-native aquatic species;
- Increase the ability to conserve and protect aquatic habitats;
- Implement measures to prevent hybridization with introduced Lahontan tui chubs;
- To the extent possible, reduce the probability of the loss of Owens tui chub populations from stochastic events; and
- Complete an approved management plan and implementing agreement that address water quantity and groundwater management with the land managers.

These recovery plan criteria do not address threats from disease; catastrophic events that may affect the Owens Basin; demographic, genetic, or environmental stochasticity; or climate change. The recovery plan identifies no recovery criteria for the Toikona lineage, as the occurrence of this lineage was unknown when the recovery plan was approved. The 5-year review (USFWS 2009) finds that none of these management goals has either not been achieved or can't be evaluated.

Data Characterization

The distribution of and threats to Owens tui chub are sufficiently well known to allow coverage of this species in the Desert Renewable Energy Conservation Plan. Missing pieces of information on this species include the lack of understanding of the Toikona lineage as far as origin, genetics, and ecophysiology (Chen et al. 2007). Additionally, the lack of management plans at each of the six existing populations has resulted in less than ideal protections for the species and a poor understanding of the population dynamics. A reintroduction plan with a specific genetic distribution of the current populations is also needed. Considering the degree of known

introgression between Lahontan and Owens tui chub (Chen et al. 2007), data on the distribution of genetically pure Owens tui chub and existing barriers is key.

Management and Monitoring Considerations

The Plan Area includes the former Cabin Bar Ranch population at Southern Owens Dry Lake. The Mule Spring population is the closest extant population, which occurs about 2 miles outside the Plan Area boundary. There are also two proposed conservation areas in the Plan Area: Black Rock and Southern Owens Dry Lake. The genetically important and distinct Toikona lineage that occurs at Mule Spring descended from a total of 24 founders from Cabin Bar Ranch and its extant population is confined to two diminutive artificial ponds at Mule Spring (Chen et al. 2007). Chen et al. (2007) have determined that the Owens tui chub lineage is more genetically distinct from the Toikona lineage than the Lahontan tui chub, which illustrates the genetic importance of the Toikona lineage. They have also determined that the Toikona lineage is suffering from low genetic variation that may be a consequence of founder effects. Specific management within the Plan Area may include development of a management plan specific to the Mule Spring population. The management plan should propose methods to secure the conservation and the management of water quantity, water quality, habitat, and aquatic predators at the existing occupied ponds at Mule Spring. It should also illustrate in detail how to create new populations for the Toikona lineage, as well as increase effective population size. This detail should include a specific standardized genetic protocol. Candidate conservation areas to be evaluated within the Plan Area for new Toikona lineage populations may include Black Rock and Southern Owens Dry Lake. Evaluation criteria may include the presence of suitable habitat and the absence of predators and the Lahontan tui chub and their hybrids. Because so little is known about the Toikona lineage, additional studies and research should be proposed, such as origin, genetics, and ecophysiology.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Owens tui chub, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 17,384 acres of modeled suitable habitat for Owens tui chub in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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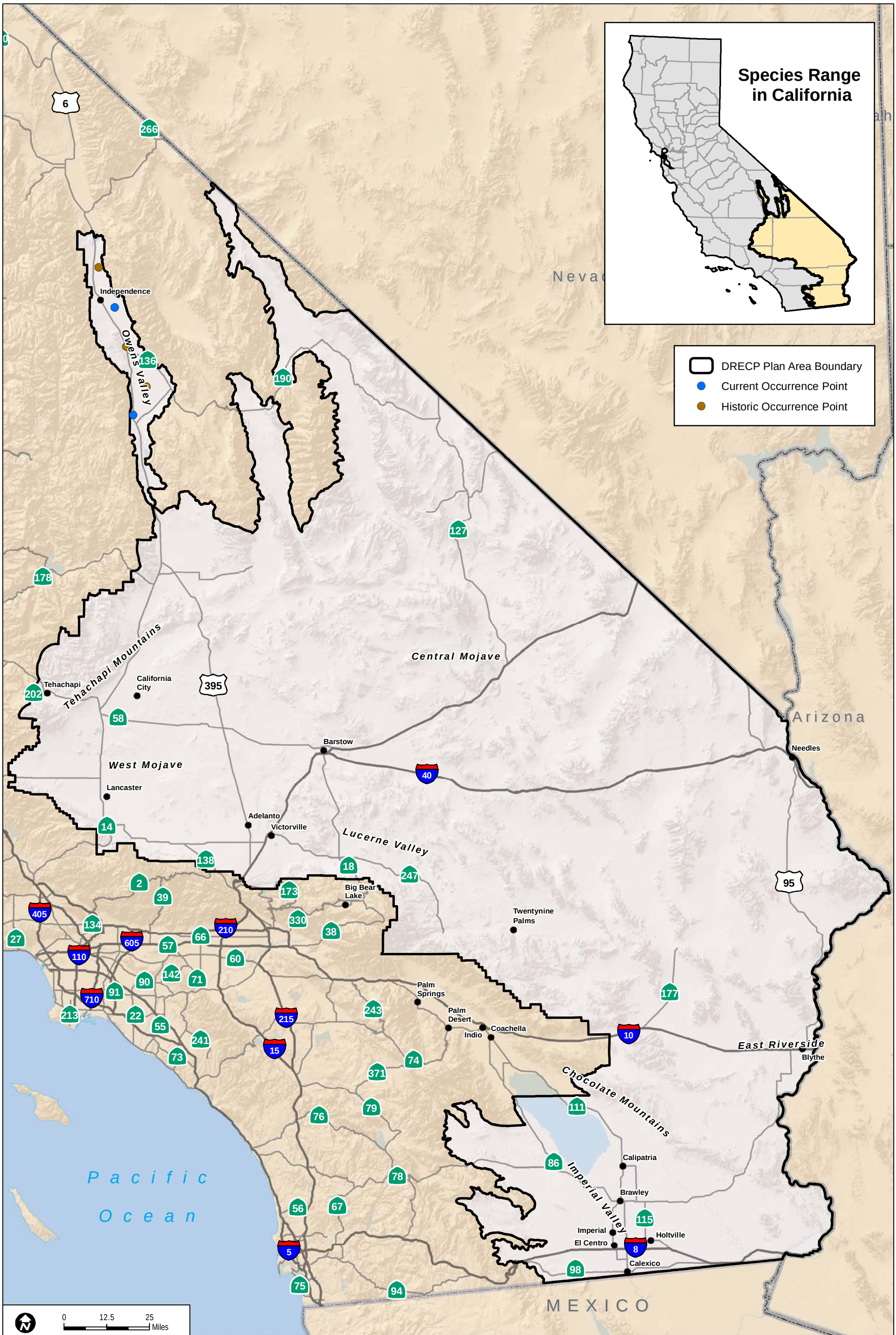

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-F04

Owen's Tui Chub Occurrences in the Plan Area

Greater Sandhill Crane (*Grus canadensis tabida*)

Photo courtesy of Brock Ortega, Dudek.

Legal Status

State: Threatened/ Fully Protected
Federal: Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

Greater sandhill crane (*Grus canadensis tabida*) is one of three subspecies of sandhill crane by the last edition of the American Ornithologists' Union Check-list of North American Birds to include subspecies (5th ed.). More recently, three additional subspecies have been recognized (Johnsgard 1983; Archibald and Meine 1996; Clements et al. 2011). Of the six subspecies, three are migratory (including *G.c. tabida*) and three are non-migratory; each of the non-migratory subspecies is listed under the federal Endangered Species Act. Subspecies boundaries in sandhill crane are significant for conservation and legal status, as well as for game management.

The three migratory subspecies, including greater sandhill crane, are separated by morphology, especially size: greater sandhill crane is the largest; Canadian sandhill crane (*G.c. rowani*) is intermediate in size; and lesser sandhill crane (*G.c. canadensis*) is the smallest (Tacha et al. 1992). However, the greater and Canadian subspecies are not fully separated. They also intergrade and apparently pair randomly at the limits of their ranges (Tacha et al. 1992; Archibald and Meine 1996). Since the recognition of the intermediate-sized Canadian subspecies, there have been several studies with varying conclusions on the limits and validity of the three migratory subspecies (summarized in Rhymer et al. 2001), and the separation of *G.c. rowani* as a distinct subspecies may not be well-grounded (Rhymer et al. 2001).

Five populations of greater sandhill crane are recognized based on morphological and geographical differences, suggesting some genetic distinctness, but these differences do not merit recognition at the subspecies level. The majority of sandhill cranes that visit the Desert Renewable Energy Conservation Plan (DRECP) area belong to the Lower Colorado River Valley (LCRV) population, but some may also be some connection with the Central Valley population (Meine and Archibald 1996).

Illustrations and descriptions of the greater sandhill crane's physical characteristics can be found in Johnsgard (1983) and Archibald and Meine (1996).

Distribution

General

Greater sandhill crane formerly occupied a much larger breeding range than it does now, ranging across the western and mid-continent from the southern portions of the western and central provinces of Canada (British Columbia, Alberta, Saskatchewan, and Manitoba) to as far south as northern California, Nevada, and Arizona, and northwestern New Mexico in the west and northern Illinois and southern Ontario, Canada in the midwest (Rhymer et al. 2001). Its Hunting and habitat loss beginning in the 1930s greatly reduced the population size and range, but has expanded in recent years. Because of interbreeding with lesser sandhill crane, the northern limits of the population are difficult to define, but the current breeding range of the greater sandhill crane now generally includes contiguous areas of Canada from British Columbia in the west to Wisconsin, Michigan and southern Ontario in the east (Rhymer et al. 2001; Tacha et al. 1992). Disjunct breeding populations occur in four areas of the western U.S.: (1) the nexus of northeastern California, southeastern Oregon and northwestern Nevada; (2) northeastern Nevada; (3) along the border region of Idaho and Wyoming north to southern Montana and south to northern Utah; and (4) northwestern Colorado (Rhymer et al. 2001; Tacha et al. 1992). Sandhill cranes winter in the southern United States and northern Mexico (Tacha et al. 1991). Wintering locations in California include the lower Colorado River and Salton Sea area, and Imperial Valley and the Central Valley (Patton et al. 2003; Rosenberg

et al. 1991 Tacha et al. 1991) (Figure SP-B10). Sandhill cranes also historically wintered abundantly at the Colorado River delta at the head of the Gulf of California in Mexico, about 80 kilometers (50 miles) south of Yuma, Arizona, and was still wintering in Sonora, Mexico in moderate numbers in recent years (Russell and Monson 1998 p. 87, as cited by Campbell, pers. comm. 2012).

Distribution and Occurrences within the Plan Area

Sandhill cranes are winter visitors to the Plan Area and have never been documented to breed in Southern California. Greater sandhill cranes that overwinter in the Plan Area belong to two populations: the Central Valley population and the LCRV population (Meine and Archibald 1996). The Central Valley population breeds in northeastern California and adjacent south–central and southeastern Oregon, and at scattered sites in southern British Columbia and on Vancouver Island. This population mainly overwinters in the Central Valley and perhaps in the Imperial Valley. The LCRV population breeds mainly in northeast Nevada and portions of adjacent states and winters in the LCRV and the Imperial Valley.

Historical

Historically, the LCRV population wintered south along the Colorado River Valley from eastern Nevada as far south as the delta in the Gulf of California (Kruse et al. 2011). Wintering greater sandhill cranes occurred “sparingly” south to the Imperial Valley, and lesser sandhill cranes also overwintered in Southern California, including the Colorado River Valley, the Imperial Valley, and the south end of the Salton Sea (Grinnell and Miller 1944).

Garrett and Dunn (1981) also stated that both greater and lesser sandhill crane subspecies overwintered in Southern California and noted that the relative abundance of the two forms is imperfectly known. They described greater sandhill crane as a regular winter visitor, with overwintering birds known from several scattered locations in the Plan Area: in the fields between Brawley and El Centro in Imperial County, in fields along the Colorado River north of Blythe and in the Cibola area in Riverside County, and in small numbers in the Needles/Topock area in San Bernardino County. Detailed

historical counts of wintering sandhill cranes in the lower Colorado River in California are provided in Appendix C of the Pacific Flyway Council's 1995 Management Plan.

There are no historical records for the greater sandhill crane in the California Natural Diversity Database (CNDDB) for the Plan Area (CDFW2013; Dudek 2013).

Recent

The current overwintering distribution in the Plan Area is similar to that described by Garrett and Dunn (1981), with several regularly used winter locations in both the Imperial Valley south of the Salton Sea and along the Colorado River. Patten et al. (2003) indicate that historically the great majority of wintering sandhill cranes in the Imperial Valley were lesser sandhill cranes and most wintering along the Colorado River were the greater subspecies, but both subspecies are known in both areas and recent relative numbers are unclear. Patten et al. (2003) also cite five records for the species at or near the north end of the Salton Sea; three in winter and one each in fall and spring.

There are no recent (i.e., since 1990) occurrence records in the CNDDB (CDFW 2013; Dudek 2013) for greater sandhill crane, but there are 16 recent occurrence records contained in the eBird database for the Plan Area for the species (the database does not include subspecies information) (Dudek 2013). These observations are primarily located south of the Salton Sea and along the lower Colorado River, with one 2011 (January) observation from Silver Lake (in Galileo Park) in California City in the western Mojave Desert (Figure SP-B10) (Dudek 2013). This small number of database occurrences, however, does not clarify the common use of the Salton Sea, Imperial Valley and lower Colorado River areas by large numbers of greater sandhill cranes in overwintering congregations. Recently, approximately 250 to 300 overwintering greater sandhill cranes were estimated to forage in privately owned grain fields south of Brawley in the Imperial Valley (Cooper 2004; Schram 2006). A recent local report describes an overwintering group of about 400 cranes foraging during the day near the intersection of Keystone and Dogwood, and roosting at night at private duck clubs in the nearby Mesquite Lake area (Kalin 2005), and this area is known to be a reliable site for

overwintering sandhill cranes (Schram 2006). Several hundred sandhill cranes currently winter in Unit 1 of the Sonny Bono Salton Sea National Wildlife Refuge (NWR) (Kruse et al. 2011). Along the lower Colorado River, sandhill cranes have been observed west of the River south Earp and just north of Blythe.

Away from the Colorado River and Salton Sea/Imperial Valley area, in addition the 2011 California City observation noted above, there are 16 records in the Plan Area published in *North American Birds* magazine for the period from 1981 through 2005 (Campbell, pers. comm. 2012). Half are in the Owens Valley, from Bishop south to Owens Lake, with the others at Desert Center (2 records), Harper Dry Lake (2), Ridgecrest (2), Death Valley (1), and near Lancaster (1). Seasonally they extend from September 11 to May 20, with 10 records in fall, 2 in winter, and 3 in spring (Campbell, pers. comm. 2012).

Natural History

Habitat Requirements

Greater sandhill cranes are found primarily in open freshwater wetlands, including shallow marshes and wet meadows (Tacha et al. 1992; Meine and Archibald 1996). They nest in moist areas at the margins of extensive wet meadows and marshes (Tacha et al. 1992). Migrating and wintering greater sandhill cranes often forage in agricultural fields, especially stubble or disked fields where grain crops have been harvested (Tacha et al. 1992). Overwintering birds in the Plan Area use irrigated pastures and croplands, grain fields, and dairy farms (Meine and Archibald 1996). Migrating and wintering birds typically use roost sites in shallow wetlands near foraging areas.

Table 1. Habitat Associations for Greater Sandhill Crane

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Freshwater wetlands	Nesting, foraging, roosting, migration staging	Primary habitat	Open areas with minimal disturbance, no or few trees, shallow water, variety of marsh and	Direct observations and surveys

Table 1. Habitat Associations for Greater Sandhill Crane

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
			vegetation types, usually with short vegetation	
Agricultural fields	Foraging in winter	Secondary habitat	Harvested / flooded agricultural fields of grain or truck crops; also irrigated pasture	Direct observations and surveys

Sources: Johnsgard 1983; Tacha et al. 1992; Meine and Archibald 1996.

Foraging Requirements

Sandhill cranes forage primarily in open, shallow freshwater wetland habitats and agricultural fields, such as irrigated pasture and harvested croplands with waste grain (Tacha et al. 1992). They are omnivorous, eating a variety of small animals and plant material that they glean from the surface or subsurface (Tacha et al. 1992). In addition, their diet varies widely depending on season and location; they are therefore able to adapt to changes in habitat and food availability to some extent. Typical native plant materials include tubers and seeds of aquatic plants. For overwintering birds, waste grain is a very important component of the diet. A wide variety of animal prey items is taken, including large invertebrates and small vertebrates such as mice, frogs, fish, and birds (summarized in Stone 2009). Cranes forage in vigilant groups in open areas where visibility is good; they are sensitive to disturbance and are easily flushed by approach, often leaving the area. For cranes foraging on agricultural fields, the level of disturbance from typical daily farm activities can be enough to disrupt foraging.

In the Plan Area, overwintering greater sandhill cranes predominantly forage in agricultural fields and irrigated pastures. Overwintering cranes near Brawley have been observed foraging in irrigated pastures of ryegrass, alfalfa, and Bermuda grass, as well as feeding on spilled grain

along railroad tracks near a grain unloading facility north of Keystone (Kalin 2005). Alfalfa and milo fields were readily used along the Colorado River (Rosenberg et al. 1991), as well as corn fields grown for waterbird forage at Cibola National Wildlife Refuge (NWR) (Oldham, pers. comm. 2012). Overwintering cranes in the Plan Area are heavily dependent for foraging throughout the winter on agricultural fields that are close to safe shallow-water wetlands for roosting at night.

Reproduction

Sandhill cranes form pair bonds that last for life, and do not breed until they reach 2 to 7 years of age (Tacha et al. 1992). Each pair maintains a breeding territory, and both male and female build a large nest of plant material typically placed in shallow water or dry land at the margin of a wetland (Tacha et al. 1992). They produce a single clutch, almost always of two eggs, and eggs are incubated for about 30 days (Tacha et al. 1992). The chicks are ready to leave the nest soon after hatching and begin feeding after about 1 day. Both parents assist in feeding the chicks. If food is limited only one chick may survive, but if the food supply is adequate, both chicks may survive. Soon after their first flight, young birds depart with their parents on the southward migration to their wintering grounds, and remain with their parents throughout the winter until they are 9 or 10 months old (Tacha et al. 1992).

Table 2. Key Seasonal Periods for Greater Sandhill Crane

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding/Fledging				X	X	X	X	X				
Migration		X	X	X					X	X		
Overwintering	X	X								X	X	X

Sources: Johnsgard 1983; Tacha et al. 1992; Meine and Archibald 1996; Schram 2006.

Spatial Activity

For the species as a whole, overwintering sandhill cranes typically arrive in Southern California during October and depart from

February through March (Schram 2006, p. 389). Spring migration for the LCRV population may begin as early as the first week of February (Pacific Flyway Council 1995; Kruse et al. 2011). Cranes depart northward and at least some stage at Lund in Nevada, where they spend a few weeks before continuing north to the breeding grounds by mid-March (Pacific Flyway Council 1995). In fall, cranes move to pre-migratory staging areas in Ruby and Lamoille Valleys in Elko County, Nevada and assemble before heading south at the end of October along the White River to their wintering grounds (Pacific Flyway Council 1995). The majority of the population overwinters at the Cibola NWR on the Arizona side of the Colorado River, with several hundred birds along the California side of the valley and in the Imperial Valley (Kruse et al. 2011). The migration route of the LCRV population is one of the shortest among the migratory sandhill cranes.

A survey of wintering birds at the major concentrations in the LCRV area in 1986 showed that 61% of cranes that had been captured and marked in the summer breeding range in Nevada were observed in the LCRV population winter range; in contrast, only 30% of the LCRV winter population has been located in the Nevada summer range (Pacific Flyway Council 1995).

Ecological Relationships

Most of the foraging and roosting sites for greater sandhill crane are on private lands used for farming and by duck clubs, and the cranes are subject to disturbance from farm activities and hunting. Collision with power lines that traverse the agricultural areas is a potential cause of injury or death for cranes flying between foraging areas. Losses to predators are rare because the birds forage in groups in open areas where visibility is good.

Population Status and Trends

Global: Increasing (Tacha et al. 1992; Meine and Archibald 1996)
State: Increasing (Central Valley population); some western populations may be declining (Meine and Archibald 1996)
Within Plan Area: Increasing (LCRV population, Kruse et al. 2011; group wintering in Imperial Valley, Kalin 2005)

The LCRV population is currently the least numerous of the migratory crane populations (Kruse et al. 2011). Aerial surveys of the major overwintering concentrations of the LCRV populations (lesser and greater) have been conducted since 1998 (at two sites in Arizona and the Sonny Bono Salton Sea NWR and Gila River), and suggest that the overall numbers are increasing at a rate of about 3% per year, from an estimated 1,900 in 1998 to 2,415 counted in 2011 (Kruse et al. 2011). However, the relative numbers of greater and lesser sandhill cranes across time is poorly known, casting uncertainty on trends for the greater sandhill crane population here.

The portion of the Plan Area total numbers overwintering at the Salton Sea NWR increased in parallel with the overall increase, from 351 in 1998 to 899 in 2011 (Kruse et al. 2011). The recruitment rate of this population is one of the lowest for sandhill cranes (Drewien et al. 1995) at 4.8% with a mean brood size of 1.14 for the periods 1973–1975 and 1989–1992 (Drewien et al. 1995). However, the most recent recruitment survey, conducted in early spring 2011, indicated a much higher rate of 9.36% (Rabe undated, cited in Kruse et al. 2011).

Threats and Environmental Stressors

The most significant current threat to the greater sandhill crane subspecies appears to be habitat loss and degradation, especially on the wintering grounds in California and Florida, the nesting areas in the Midwest, and migration stopovers, especially the Platte River (Meine and Archibald 1996).

Several specific habitat issues of concern for the LCRV population winter grounds have been identified: (1) a shortage of good roosting sites near foraging areas with grain fields; (2) lack of management and control over agricultural crops that provide winter foraging; (3) destruction of roost sites by past and proposed dredging and channelization projects along the Lower Colorado River: and (4) conversion of croplands from grain to crops that do not provide good foraging for cranes, such as alfalfa and cotton (Pacific Flyway Council 1995). In addition, potential impacts of water transfers and fallowing of agricultural areas in both Imperial Valley and lower Colorado River Valley could have critical impacts on winter grounds (Campbell, pers. comm. 2012).

Conservation and Management Activities

The greater sandhill cranes overwintering in Southern California (the LCRV population) have not been hunted since 1918; however, in 2007 the U.S. Fish and Wildlife Service completed an Environmental Assessment on proposed hunting regulations for this population, and in 2008 proposed a small allowable harvest of 30 birds in years when the wintering population numbers exceeded 2,500; the proposed harvest is guided by a cooperative management plan (Pacific Flyway Council 1995). No cranes have been harvested yet because the population remains below the 2,500-bird threshold (Kruse et al. 2011).

The exact breeding location of about 70% of the wintering LCRV population is uncertain, and the Arizona Game and Fish Department is currently investigating movement patterns and breeding locations by placing satellite transmitters and alphanumeric bands on wintering birds so their movements can be tracked (Ingraldi and Frary 2010).

The Pacific Flyway Management Plan for the LCRV population of greater sandhill crane (Pacific Flyway Council 1995) provided a series of management recommendations grouped into several categories: habitat, environmental education and law enforcement, inventories, and research. The habitat recommendations were focused on the nesting and stopover sites, in addition to the wintering grounds. Winter roost sites were identified for protection and acquisition, including two key sites southeast of Brawley: the D & K Duck Club and Osterkamp Farms.

To address the shortage of foraging habitat close to suitable roost sites, at Cibola NWR on the Arizona side of the Colorado River, where the largest concentration of the LCRV population spends the winter, additional foraging has been provided by planting corn crops near suitable roost sites, and this has proved successful in maintaining and increasing the crane numbers there.

Data Characterization

There are three important areas of information uncertainty at this time. First, the uncertainty over the breeding range of about 70% of the LCRV wintering population has implications for the overall

management of this population and adjacent populations. However, despite the uncertainty over their summer range, the LCRV population consistently winters in the Plan Area, and, assuming the population is not limited entirely by factors away from the winter grounds, conservation measures implemented under the DRECP would benefit the population wintering in the Plan Area.

Second, there is ongoing uncertainty about the relative proportions of the lesser and greater sandhill crane subspecies, both in the Imperial Valley and along the lower Colorado River, masking population trends in the LCRV population of greater sandhill crane. Depending on limiting factors present in the two populations, it is also possible that competition with lesser sandhill crane could pose some degree of threat to the LCRV greater sandhill crane population (Campbell, pers. comm. 2012).

Third, there is uncertainty regarding the effect on habitat of changes in agricultural practices as a result of changes in water availability on wintering grounds. Specific issues include political developments, such as water transfers, the effect of climate change, and the potential interaction of these two issues (Campbell, pers. comm. 2012).

Management and Monitoring Considerations

Monitoring sandhill crane numbers is relatively straightforward because the cranes are large, diurnal, gregarious birds that forage in open habitats. For at least the early part of the winter, young birds can be distinguished from adults, allowing annual recruitment to be quantified and monitored. Annual surveys using consistent methods are ongoing and provide a reasonably accurate tracking of species numbers and trends.

Though not critical within a single year, it will be important over time to adequately distinguish the LCRV population of greater sandhill cranes from the lesser sandhill cranes wintering in the Plan Area. This will prevent masking of changes in the numbers of greater sandhill cranes by data for the other subspecies (Campbell, pers. comm. 2012). Potential techniques include monitoring the cranes vocalizations (Jones and Witt 2012) or more traditional trapping of cranes or training of observers.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for greater sandhill crane, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 638,431 acres of modeled suitable habitat for greater sandhill crane in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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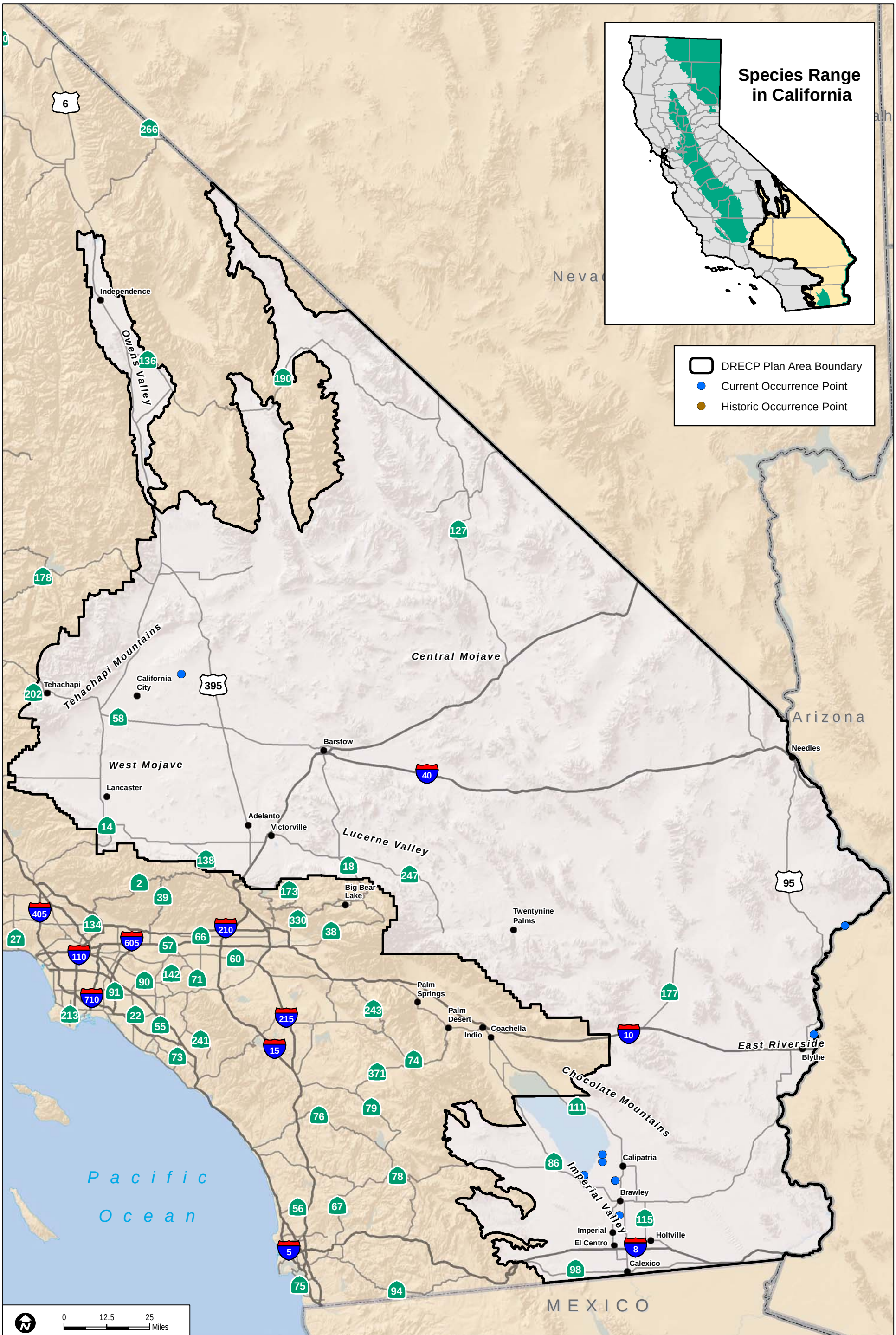

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B08

Greater Sandhill Crane Occurrences in the Plan Area

Least Bell's Vireo (*Vireo bellii pusillus*)

Photo courtesy of Brock Ortega, Dudek.

Legal Status

State: Endangered in California.
Federal: Endangered.
Critical Habitat: Designated (59 FR 4845–4867)
Recovery Planning: U.S. Fish and Wildlife Service (USFWS) 1998.
Notes: The species Bell's Vireo is also listed as a Bird of Conservation Concern by the USFWS within the Mojave Desert Bird Conservation Regions (BCR) (USFWS 2008).

Taxonomy

There are four recognized subspecies of Bell's vireo (*Vireo bellii*) including *V. b. belli; V. b. medius; V. b. arizonae*; and *V. b. pusillus*, the least Bell's vireo (AOU 1998). While all subspecies are similar in appearance, least Bell's vireo is mostly gray above and pale below, while easternmost birds are greenish above and yellowish below. Southwestern subspecies are intermediate in plumage characteristics. Descriptions of the species' physical characteristics, behavior, and distribution are provided in a variety of field guides (e.g., Peterson 1990; Sibley 2000; National Geographic 2002).

Distribution

General

Bell's vireo is a migratory species that breeds in North America. Least Bell's vireo breeds in central and southern California, and northwestern Baja California. In California, breeding takes place through coastal Santa Barbara County to San Diego County, San Bernardino, Riverside, and Inyo Counties (USFWS 2006). A few isolated least Bell's vireo have been observed in Kern, San Benito, Monterey, and Stanislaus Counties since the species was listed but these counties have not supported any sustained populations.

In California, the historic range of least Bell's vireo has severely contracted. Historically, the breeding range of the least Bell's vireo subspecies was widespread throughout California, including the Sacramento and San Joaquin Valleys (Grinnell and Miller 1944), Sierra Nevada foothills, and in the Coast Ranges from Santa Clara County south to approximately San Fernando, Baja California, Mexico (USFWS 1998). Populations were also known from the Owens Valley, Death Valley, and at scattered oases in the Mojave Desert (Kus et al. 2010; USFWS 1998). At the time of listing in 1986, over 99% of the least Bell's vireo population was found south of Santa Barbara County (USFWS 2006).

The least Bell's vireo subspecies overwinters primarily along southern Baja California (Kus 2002a) while the Arizona Bell's vireo subspecies overwinters primarily in northwestern Mexico (Kus, pers. comm. 2012) (Figure SP-B02).

Breeding habitat for all subspecies of Bell's vireo generally consists of dense, low, shrubby vegetation, (early successional stages) in riparian areas, and mesquite brushlands, often near water in arid regions (Kus et al. 2010). Bell's vireo winter in both riparian and upland vegetation but in habitats more widely distributed away from water. Least Bell's vireo winters in willow riparian habitat, arroyo scrub vegetation and hedgerows in coastal drainages (Kus et al. 2010).

Distribution and Occurrences within the Plan Area

Historical

In California by the early 1980's, least Bell's vireo was extirpated from most of its historic range, with small populations remaining in coastal southern California (U.S. Fish and Wildlife Service 1998). There are four historical (i.e., pre-1990) occurrences of least Bell's vireo in Inyo County in the northern portion of the Plan Area and in the southern portion of the Plan Area in and west of Joshua Tree National Park (Dudek 2013).

There are also three historical occurrences for Bell's vireo where the species occurrence in the database is not identified to subspecies (Dudek 2013). These observations were in the Shadow Valley area west of the Mesquite Mountains, near Shoshone, and near Furnace Creek (Figure SP-B02).

Recent

At the time of its federal listing, least Bell's vireo had been extirpated from most of its historic range, and numbered just 300 pairs statewide (Kus 2002a; USFWS 1998). Due to extensive habitat protection and cowbird control programs, the least Bell's vireo is increasing throughout southern California, with a tenfold increase in the recorded population since its listing in 1986 (USFWS 2006) and a recent colonization of the San Joaquin River in Stanislaus Co. (Howell and Dettling 2009; see Conservation and Management Activities). However, least Bell's vireo has not yet meaningfully recolonized its historical breeding range in the Sacramento valley (USFWS 2006). Breeding pairs have been observed in the Counties of Monterey, San Benito, Inyo, Santa Barbara, San Bernardino, Ventura, Los Angeles, Orange, Riverside, and San Diego, with the highest concentration in San Diego County along the Santa Margarita River (USFWS 2006).

There are 29 recent occurrence records of least Bell's vireo in the Plan Area in the following areas: near Lancaster and Palmdale, north of Hesperia, north of Victorville, southwest of Yucca Valley, along Carrizo Creek in Anza Borrego Desert State Park, and along Owens River (Figure SP-B02) (CDFW 2013; Dudek 2013).

There are 10 recent occurrences for Bell's vireo that are not identified to subspecies in the following areas: two occurrences west of Pearsonville in the southern Sierra foothills, two occurrences in the Amargosa River area, one occurrence south of the Salton Sea, and five occurrences in the Morongo Valley area (Dudek 2013).

Natural History

Habitat Requirements

Bell's vireo is a neotropical migrant that breeds in the summer in riparian scrub (Table 1). Least Bell's vireo is largely associated with early successional cottonwood-willow and is known to nest in riparian woodlands dominated by willow (Kus et al. 2008) and Fremont cottonwood (*Populus fremontii*) (Kus 2002a). Suitable willow woodlands are typically dense with well-defined vegetative strata or layers. The most critical structural component of nesting habitat in

California is a dense shrub layer 2 to 10 feet aboveground (Goldwasser 1981; Franzreb 1989; Brown 1993). Bell's vireo is usually found along drainages or elsewhere near water, including ponded surface water or where moist soil conditions occur (Rosenberg et al. 1991), especially in arid environments (Szaro and Jakle 1982). Kus and Miner (1998) also stated the importance to least Bell's vireo of non-riparian habitats within and adjacent to floodplains for foraging and other activities. In arid environments, surface water appears to be an important element in least Bell's vireo habitat (Kus et al. 2010).

Table 1. Habitat Associations for Least Bell's Vireo

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Data
Riparian woodland	Breeding, foraging	Primary	Typically riparian woodland dominated by willow shrubs, mesquite understory, and other thick understory vegetation, including tamarisk	Goldwasser 1981; USFWS 1998; Kus et al. 2010
Riparian scrub	Breeding, foraging	Primary	Typically riparian scrub dominated by willow, mesquite understory and other thick vegetation	Goldwasser 1981; USFWS 1998; Kus et al. 2010
Mesquite Woodlands	Breeding, foraging	Primary	Historically widespread in mesquite forests , especially in riparian areas	Kus et al. 2010

Foraging Requirements

Individuals may forage in woodlands or scrub habitat near nesting habitat, concentrated in lower to mid-canopies, especially when actively nesting (Kus et al. 2010; USFWS 1998). Least Bell's vireo has shown preferences for black willow (*Salix gooddingii*) relative to its

cover in territories (Miner 1989; Kus et al. 2010). Least Bell's vireos also forage in upland vegetation adjacent to riparian corridors particularly late in the season (Gray and Greaves 1984; Kus and Miner 1998; Salata 1983). During the winter, least Bell's vireo use willow riparian habitat, arroyo scrub vegetation, and hedgerows in coastal drainages (Kus et al. 2010).

Reproduction

Breeding least Bell's vireos begin arriving on their breeding grounds in late March and begin nesting in early April (Table 2) (Kus 2002b). Individuals may remain on the breeding grounds into early October, but nesting is typically finished by the end of July (Kus 1999). Most pairs are monogamous during the breeding season (Kus et al. 2010). Reproduction is significantly affected by brown-headed cowbird nest parasitism (see Ecological Relationships below). In addition to nest loss to parasitism, some nests fail due to other causes, including precipitation damage to nest or supporting vegetation or effects from human or animal activity, desiccation of supporting host plant, infertile or otherwise unviable eggs (Kus et al. 2010), and nest predation by a range of species including western-scrub jays (*Aphelocoma californica*), snakes, Cooper's hawk (*Accipiter cooperii*) and raccoons (*Procyon lotor*) (USFWS 1998; Kus et al. 2008).

Table 2. Key Seasonal Periods for Least Bell's vireo

	Jan	Feb	Mar	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding				✓	✓	✓	✓					
Migration		✓	✓	✓				✓	✓	✓		
Wintering	✓	✓								✓	✓	✓

Sources: Brown 1993; Kus 1999, 2002b.

Spatial Behavior

Little is known about the migratory routes of this species (Table 3). Most individuals have left the United States by early October (Brown 1993). During spring migration, adults return to their breeding grounds in mid-March to mid-April (Brown 1993; Kus et al. 2010). In California and Arizona, males arrive on breeding areas 1 to 2 weeks before

females (Kus et al. 2010). The species' migratory behavior is poorly known, although it is thought to be chiefly a nocturnal migrant. Home range and movement during the breeding season is limited to areas within dense riparian corridors. Territories are often linear in nature, following the stream course.

Table 3. Movement Distances for Least Bell's Vireo

Type	Distance/Area	Location of Study	Citation
Home Range	0.6 to 0.9 ha	California	Newman 1992
	0.5 to 4 acres (0.2 to 1.6 ha)	California	Gray and Greaves 1984
	0.7 ha	California	Collins et al. 1989
	0.7 to 1.1 ha	California	Kus 1991, 1992, 1993
Dispersal	33 feet on day 1 to 330 feet on day 5	Indiana	Hensley 1950
	100 to 200 feet on day 14		Nolan 1960
Migration	From breeding grounds in U.S. and overwinters in southern Baja California and northwestern Mexico	California and Arizona	Kus et al. 2010

Ecological Relationships

For breeding, this species is dependent on dense riparian corridors, typically along watercourses. Scrub habitats adjacent to these watercourses are also important to the success of the species because they provide foraging opportunities as well as protection for nesting habitat.

Brown-headed cowbirds have decimated least Bell's vireo populations throughout its breeding range through nest parasitism. Dense riparian breeding habitat that is surrounded by agricultural lands or developed areas could facilitate brown-headed cowbird abundance and lower the breeding success of riparian nesting species such as the least Bell's vireo.

In California, more than a third of least Bell's vireo nests from the late 1920s through the 1980s contained cowbird eggs (Goldwasser et al. 1980). Since widespread implementation of cowbird trapping, over

the last 25 years, parasitism rates have dropped substantially and Bell's vireo nesting success has increased dramatically (see Conservation and Management Activities) (Griffith and Griffith 2000; Kus 1999; Kus and Whitfield 2005).

Cowbirds typically parasitize vireo nests during the egg-laying period and female cowbirds often remove or destroy vireo eggs. Adult Bell's vireos will attack female cowbirds to defend their nests (Mumford 1952; Budnik et al. 2002; Sharp and Kus 2004). In some instances Bell's vireo will abandon nests parasitized by cowbirds. A study in California showed that vireos continued to incubate 3 of 3 videotaped nests in which cowbirds laid eggs (Sharp and Kus 2004).

Population Status and Trends

Global: Declining (Kus 2002b; NatureServe 2005; Kus et al. 2010)
State: Recent evidence of range extensions and population increase (USFWS 2006)
Within Study Area: Unknown, may be increasing

Least Bell's vireo was described as common or abundant in the late 1800s and early 1900s (USFWS 1998). In California, the precipitous decline in numbers has been due to loss and degradation of riparian habitat, and the expansion in range of the brown-headed cowbird (USFWS 1998).

By 1986, the least Bell's vireo population had declined to an estimated 300 pairs, with the majority occurring in San Diego County (USFWS 1998; Kus 2002a). In 2006, the statewide population in California numbered approximately 3,000 territorial males (U.S. Fish and Wildlife Service 2006).

The USFWS records show a tenfold increase in the least Bell's vireo population since its listing under the federal ESA in 1986, from 291 to 2,968 known territories, with "tremendous" growth of the vireo populations in specific areas in San Diego and Riverside counties and lower but still significant growth in Orange, Ventura, San Bernardino, and Los Angeles counties (USFWS 2006). However, there have been significant declines in least Bell's vireo populations in Santa Barbara County since its original listing, while Kern, Monterey, San Benito, and

Stanislaus Counties have not supported any sustained populations (USFWS 2006).

Threats and Environmental Stressors

Historic loss of riparian habitat associated with agricultural practices, urbanization, and exotic plant invasion has contributed to decline of the species (USFWS 2006). Loss of breeding habitat due to water source alteration (e.g., flood control and channelization), urbanization, and livestock grazing also threatens the species. In addition, nest parasitism by the brown-headed cowbird has greatly reduced nest success throughout most of its breeding range and has been suggested as a primary cause for decline throughout California. A recent study found that vireo productivity increased by one young for each 30% decrease in nest parasitism (Kus and Whitfield 2005). An increase in cowbird abundance is propagated by particular land-use practices (e.g., residential development, agriculture, grazing) on lands adjacent to breeding habitats (Kus 1999; NatureServe 2005). In urbanized areas, where habitat is fragmented and breeding habitat lacks buffers, nest predation may also increase due to meso-predator release and the addition of non-native predators such as domestic or feral cats (USFWS 2006). The exotic Argentine ant (*Linepithema humile*) also has been noted as a nest predator (Peterson et al. 2004).

Other threats to this species' habitat include urban and suburban development on floodplains, the presence of large areas of invasive plants, such as tamarisk and giant reed (*Arundo donax*), and off-road vehicular activity (Wildlife Action Plan Team 2006). Also, flood control projects and grazing have destroyed much of the western nesting habitat (NatureServe 2010).

Conservation and Management Activities

Near the Plan Area, the least Bell's vireo is covered by the Coachella Valley Multiple Species Habitat Conservation Plan (MSHCP), which aims to conserve habitat of covered species. One of the goals of the Coachella Valley MSHCP is to ensure species persistence in the Plan Area by protecting and managing riparian habitat, controlling invasive plants, such as tamarisk, and controlling brown-headed cowbird populations, when necessary. The MSHCP will protect and manage in

perpetuity 1,282 acres of modeled breeding habitat and 19,301 acres of migratory habitat. The plan will also establish 44 acres of Sonoran cottonwood-willow riparian forest.

Various integrated natural resource management plans (INRMPs), developed as part of compliance under the Sikes Act Improvement Act of 1977, have successfully contributed to vireo conservation, including the 2001 INRMP for Camp Pendleton, which includes management actions such as cowbird trapping, which has improved population numbers in the short-term (USFWS 2006).

Bell's vireo is included in the Partners in Flight North American Landbird Conservation Plan (Rich et al. 2004), where it is designated as a Watch List species that warrants immediate action. Additionally, the species is on the USFWS list of Birds of Conservation Concern 2008 (USFWS 2008).

Throughout California, the listing of least Bell's vireo prompted protection of existing habitat, creation and planting of riparian habitat, the restoration of degraded habitat, largely through the removal of invasive exotic species such as giant reed, and widespread cowbird control through annual trapping of cowbirds from riparian habitats and nest manipulation to remove cowbird eggs from vireo nests (Beezely and Rieger 1987; U.S. Fish and Wildlife Service 1998; Griffith and Griffith 2000; Kus 2011). Relocation of feedlots, dairies, and stables away from riparian areas, and reduction of grazing in riparian areas is also recommended (USFWS 1998).

Reintroduction of Bell's vireos to areas within their historical California range has been considered but not done (Franzreb 1989). Sharp and Kus (2006) propose managing for dense understory vegetation, particularly willows, to reduce parasitism risk for nesting vireos.

Data Characterization

In general, there is a good deal of information regarding least Bell's vireo in the Plan Area. However, least Bell's vireo is highly mobile and can occur unexpectedly in new areas far from known breeding areas. Particularly, given that the species' range is expanding and population numbers are growing, continued survey work that seeks to document species presence over time is necessary.

Management and Monitoring Considerations

Bell's vireo is dependent on riparian vegetation, so management actions that improve riparian habitat will likely benefit the species. Cowbird control has been shown effective in reducing parasitism and increasing nest success throughout the vireo's range (Kus 1999, 2002b; Griffith and Griffith 2000; Morrison and Averill-Murray 2002; Kus and Whitfield 2005; Kosciuch and Sandercock 2008). A 2-year study in the Colorado River Valley of Arizona, showed that the parasitism rate and incidence of multiple cowbird eggs were significantly higher on untrapped reference plots than on treatment (trapped) plots, while success rate of nests was higher in treated plots than in the reference plots (Morrison and Averill-Murray 2002). However, Kus and Whitfield (2005) warn of using cowbird control as a long-term management tool as it makes the species' success dependent on human intervention.

Kus and Whitfield (2005) recommend practices emphasizing habitat restoration and the maintenance of natural processes on which the species depend. For example, removal of tamarisk from existing riparian areas (if replaced by native riparian habitat) would enhance habitat for least Bell's vireo and other riparian birds. Large-scale efforts to remove giant reed from drainages, such as those along the Santa Margarita River (Lawson et al. 2005) and Santa Ana River in southern California have been successful in facilitating re-establishment of native vegetation and subsequent colonization by least Bell's Vireo (SAWA 2013).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for least Bell's vireo, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 298,231 acres for least Bell's vireo in the Plan Area. Appendix C includes figures showing the modeled suitable habitat in the Plan Area.

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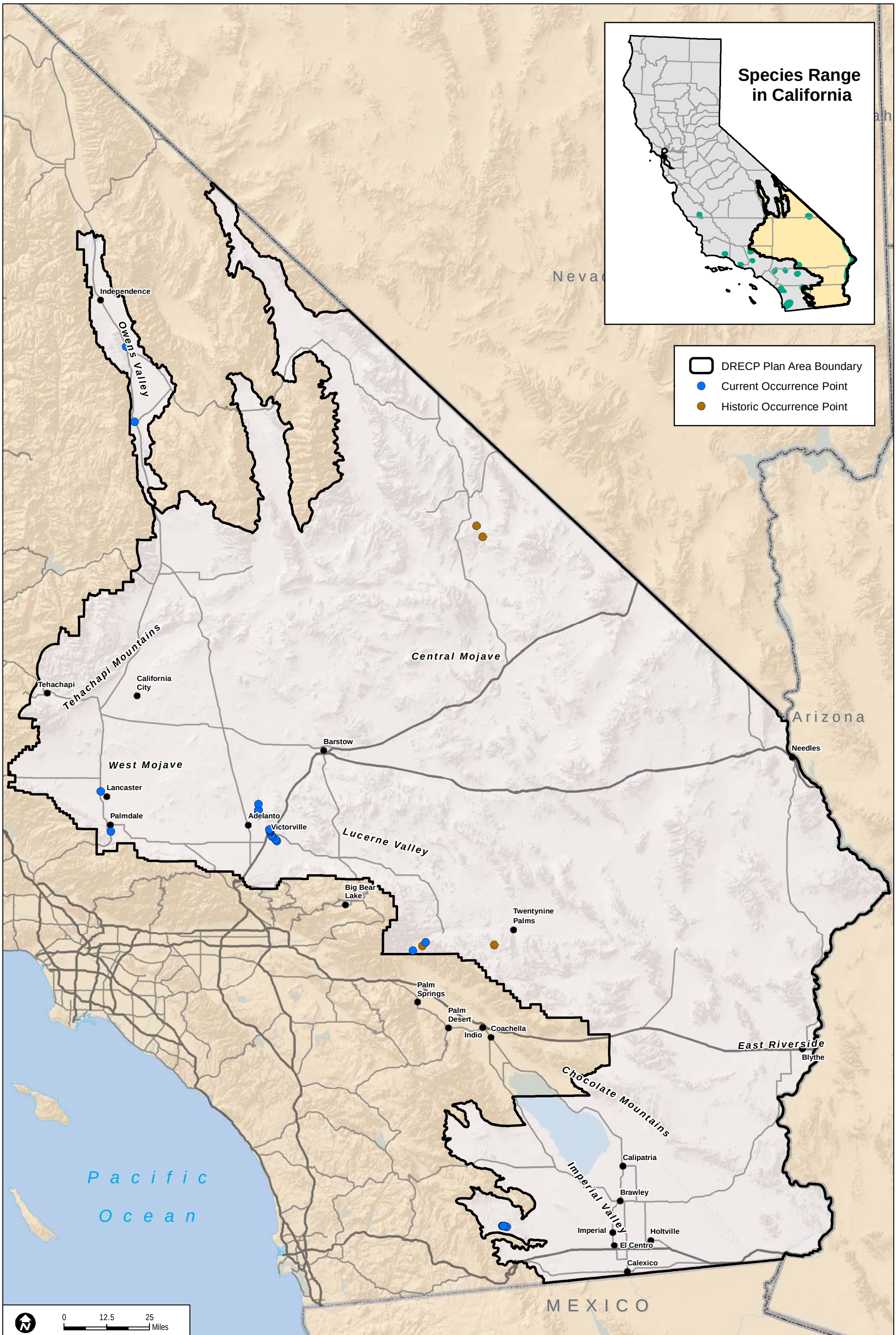

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B01

Least Bell's Vireo Occurrences in the Plan Area

Mountain Plover (*Charadrius montanus*)

Legal Status

State: Species of Special Concern
Federal: U.S. Fish and Wildlife Service Bird of Conservation Concern, Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A
Notes: Proposed listing as threatened species withdrawn on May 12, 2011 (76 FR 27756–27799).

Taxonomy

Mountain plover (*Charadrius montanus*) is a bird in the order Charadriiformes and the Charadriidae family. This species was formerly placed in a monotypic genus (Eupoda) (Garrett, pers. comm. 2012). There are no recognized subspecies of mountain plover (76 FR 27756–27799). The closest relatives to the mountain plover appear to be the Asiatic species *Charadrius asiaticus* (Caspian plover) and *C. veredus* (Oriental plover) (Garrett, pers. comm. 2012).

Distribution

General

Mountain plover occurs from Canada (AB, SK) south through the United States (AZ, CA, CO, KS, MT, ND (extirpated), NE, NM, NN, OK, SD (extirpated), TX, UT, and WY) and into Mexico. In California, where most birds winter, the mountain plover is known in the following counties: Riverside, Orange, Santa Barbara, Madera, Mono, San Bernardino, San Diego, San Joaquin, Humboldt, Kings, Monterey, Colusa, Fresno, Imperial, Kern, Los Angeles, Merced, San Benito, San Luis Obispo, Solano, Stanislaus, Tulare, Ventura, and Yolo (NatureServe 2010; Knopf and Wunder 2006).

Distribution and Occurrences within the Plan Area

Historical

In California, the historical wintering range for mountain plover included low elevation interior valleys and plains. The range extended from the southern Sacramento Valley and the inner San Francisco Bay area south to the southern coastal slope and east to the Imperial Valley. According to sources from 1944 and 1957, in the southern deserts, mountain plover historically occurred near Indio in Riverside County, at Brawley and Pilot Hill in Imperial County, and Needles in San Bernardino County (Hunting and Edson 2008).

There are 11 historical (i.e., pre-1990) occurrence records for mountain plover in the Plan Area (CDFW 2013; Dudek 2013). The majority of these occurrences are located east of Lancaster and north of Palmdale, in the southwest corner of Edwards Air Force Base, in the Harper Lake area, and at the southern end of the Salton Sea (Figure SP-B11).

Recent

In California, mountain plovers continue to occupy the same broad regions in which they have historically occurred, although they no longer winter on the Channel Islands or the eastern fringes of the San Francisco Bay area (Hunting and Edson 2008). In the southern desert region, mountain plovers winter in the Antelope Valley; western Mojave Desert, near Harper Dry Lake; the Imperial Valley; and near Blythe in the lower Colorado River Valley (Hunting and Edson 2008).

Within the Plan Area, there are 61 recent (i.e., since 1990) documented occurrences south of or along the eastern edge of the Salton Sea, near Palmdale, west of Lancaster, and in the Harper Lake area (Figure SP-B11) (CDFW 2013; Dudek 2013).

Natural History

Habitat Requirements

Although mountain plover is categorized as a shorebird, it is not actually associated with margins of freshwater or marine estuaries,

and despite its name, mountain plovers do not actually nest in the mountains (Table 1; 76 FR 27756–27799; McGaugh 2006). In California, mountain plovers primarily winter on fallow and cultivated agricultural fields, but also use grasslands and grazed pastures (76 FR 27756–27799). Audubon (2011) observed wintering mountain plovers in five habitat types: grassland, alfalfa, lettuce, beach, and in bare dirt or recently plowed fields. Alkali playa is an important habitat type in composition, structure, and location (County of Riverside 2003). In the Imperial Valley, where there is the largest known concentration of wintering plovers, preferred foraging habitats include harvested alfalfa and Bermuda grass fields that have been grazed by domestic sheep and Bermuda grass fields, wheat, and other grass fields that have been burned post-harvest (Knopf and Wunder 2006; Molina 2011; Molina 2012). Molina (2011, 2012) also demonstrates the importance of bare plowed or furrowed agricultural fields in the Imperial Valley for mountain plovers. During migration, mountain plovers likely use habitats similar to their breeding and wintering habitats (76 FR 27756–27799). Mountain plover also appear in very small numbers in coastal estuaries in Fall migration, even though they do not winter in such habitats (Garrett, pers. comm. 2012). Mountain plovers prefer areas with heavy, saline/alkaline, clay soils (BLM 2002, p. N-8; see Table 1).

Table 1. Habitat Associations for Mountain Plover

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Fallow and cultivated agricultural fields, burned grass fields, grasslands, alkali playa, and grazed pastures	Winter	Wintering	Short vegetation with some bare ground	76 FR 27756–27799; McGaugh 2006; County of Riverside 2003

Mountain plover breeding habitats are similar to those used for wintering. Suitable breeding habitat for mountain plover includes disturbed prairie or semidesert habitats at high elevations, from 2,000 to 8,500 feet (76 FR 27756–27799; McGaugh 2006; Knopf and Wunder 2006). This species occupies open, flat lands or sparsely

vegetated areas, including xeric shrublands, short-grass prairie, and barren agricultural fields. Grassland habitats where mountain plover is found often have a history of disturbance by burrowing rodents, such as prairie dogs (*Cynomys* spp.), native herbivores, or domestic livestock (76 FR 27756–27799).

Mountain plover breeding sites require short vegetation with some bare ground. Breeding habitats for mountain plover include short- and mixed-grass prairie, prairie dog colonies, agricultural lands, and semidesert areas (76 FR 27756–27799). Typical disturbances in grasslands include disturbances from prairie dogs, cattle grazing, fire, or farming. Although these forms of disturbance are usually required in grassland habitats, breeding sites in semidesert environments may persist without these forms of disturbance (76 FR 27756–27799).

Foraging Requirements

Mountain plovers feed on ground-dwelling or flying invertebrates found on the ground (76 FR 27756–27799). Their diet primarily consists of beetles, crickets, and ants, though mountain plover diets are diverse and differ greatly by location (76 FR 27756–27799; McGaugh 2006). Mountain plovers feed opportunistically as they encounter prey (76 FR 27756–27799). Foraging behavior consists of short runs and stops in which prey are captured with a lunge at the end of a short, quick run (76 FR 27756–27799; McGaugh 2006). On wintering grounds, mountain plovers also forage by probing into cracks of dried loamy soils (Knopf and Wunder 2006).

Mountain plovers forage in large areas of dry, disturbed ground or areas of short (less than 2 centimeters [0.79 inch]) vegetation with patches of bare ground. Prey is more abundant on prairie dog towns than adjoining habitats (Knopf and Wunder 2006).

Reproduction

Mountain plovers return north to their breeding sites in the western Great Plains and Rocky Mountain states in spring. Males defend territories shortly after arrival at the breeding grounds (76 FR 27756–27799). Generally monogamous, mountain plovers form pairs and begin courtship on arrival at the breeding grounds as well. In

Colorado, mountain plovers lay eggs between late April and mid-June in a simple ground scrape nest (Table 2; 76 FR 27756–27799).

Mountain plovers nest using what has been described as the ‘‘rapid multiclutch system,’’ which may increase their breeding success in the face of predation. Typically, the female produces two clutches with three eggs each at different nest sites. The male incubates one nest site and the female incubates the other. They may renest if nests or broods are lost early in the breeding season (76 FR 27756–27799). Each pair can make up to four attempts per year to raise a brood, but only one brood is raised per adult each season (76 FR 27756–27799; Knopf and Wunder 2006).

Mountain plovers incubate for 29 days on average, and young fledge at approximately 33 to 34 days (76 FR 27756–27799; Knopf and Wunder 2006). Mountain plovers can breed their first spring (76 FR 27756–27799).

Table 2. Key Seasonal Periods for Mountain Plover

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding				X	X	X						
Migration						X	X	X	X	X		
Wintering	X	X	X							X	X	X

Source: 76 FR 27756–27799; Knopf and Wunder 2006

Spatial Behavior

In late summer and early fall, mountain plovers migrate south across the southern Great Plains to Texas, New Mexico, and Mexico. Several then travel west to California (Table 3). In California, fall migrants generally arrive in the north by mid-September and in the south by mid-October (Knopf and Wunder 2006). Patten et al. (2003) indicate that the earliest date for migrating mountain plovers to arrive at the Salton Sea is August 24 but that the majority of the birds return to this area in late September. Most birds depart this area in March, with the last birds leaving by March 31. In the Antelope Valley, peak numbers of this species occur from late October to early March while peak

numbers of this species occur from late October to mid-March in the western Mojave Desert in Los Angeles County (Garrett, pers. comm. 2012). During spring migration in early March, mountain plovers travel quickly from their wintering sites to their breeding sites, arriving in eastern Colorado by mid-March and in Montana by mid-April (76 FR 27756–27799). In California, wintering mountain plover movement patterns are highly variable with some birds moving more than 34 miles in one week (76 FR 27756–27799).

Table 3. Spatial Behavior by Mountain Plover

Type	Distance/Area	Location of Study	Supporting Information
Brood home range	143 acres	Colorado	76 FR 27756–27799
Dispersal	8.1 miles for males and 6.3 miles for females	Montana	76 FR 27756–27799

Ecological Relationships

Most egg and chick losses are to predators (County of Riverside 2003). Birds, mammals, and reptiles, including prairie falcon (*Falco mexicanus*) and kit fox (*Vulpes macrotis*), are known to predate mountain plover eggs and/or chicks (McGaugh 2006).

Historically, winter areas in California supported tule elk (*Cervus elaphus nannodes*), pronghorn (*Antilocapra americana*), and kangaroo rat (*Dipodomys* spp.) (McGaugh 2006). In the Carrizo Plain, winter habitat availability is currently correlated with livestock grazing and precipitation; mountain plovers prefer dry areas that are heavily grazed. Annual climatic variability and abundant rainfall, in particular, alter field conditions, which can reduce mountain plover use of traditionally occupied wintering sites. In the Imperial Valley, mountain plover became virtually absent from cultivated fields during the rainy winter of 2004 to 2005 (76 FR 27756–27799).

Mountain plovers favor plowed or recently harvested agricultural fields and habitats that have been burned because these disturbances create the necessary sparse conditions (BLM 2002, p. N-8; 76 FR 27756–27799).

Mountain plovers prefer areas with abundant mammalian burrows (BLM 2002, p. N-8). They tend to be associated with giant kangaroo rat (*Dipodomys ingens*) colonies, especially when wet years produce tall vegetation elsewhere (76 FR 27756–27799).

Population Status and Trends

Global: Increasing (76 FR 27756–27799)
State: Same as above
Within Plan Area: Same as above

From 2004 to 2007, the International Union for the Conservation of Nature (IUCN) listed mountain plover as ‘‘vulnerable,’’ a higher level of concern than ‘‘near threatened.’’ However, higher rangewide population estimates have emerged prompting IUCN to change its rating accordingly.

From 1966 to 1993, Breeding Bird Survey (BBS) data indicate a decline rate of 3.7% per year. Although the BBS survey routes are not distributed evenly within the species’ habitat, the decline rate indicates reduction in the population during that 25-year period by approximately two-thirds (Knopf and Wunder 2006). Until 2006, a rangewide mountain plover population estimate provided by the U.S. Shorebird Conservation Plan was increased from 9,000 to 12,500 (76 FR 27756–27799).

Although wintering mountain plover populations in California appear to have experienced a significant decline over previous decades, more recent wintering numbers, from 2000 onward, have not shown a similar trend. In 2007, 4,500 mountain plover were recorded in the

Imperial Valley, which exceeded statewide survey counts of mountain plover from 1994, and 1998 through 2002. A statewide survey over 5 days in January 2011 recorded 1,235 mountain plover, which is considerably fewer than found in previous statewide surveys or recent Imperial Valley surveys. In late 2010, unusually wet conditions due to heavy rains may have influenced the relatively low number of mountain plover in California (76 FR 27756–27799).

Threats and Environmental Stressors

Mountain plovers are threatened by loss and degradation of breeding and wintering habitat, predation, severe weather conditions during nesting/fledging, and direct persecution by humans (McGaugh 2006).

Habitat loss and degradation appear to be the main factors contributing to mountain plover population declines (Hunting and Edson 2008). The reduction of short-grass prairie by conversion to agriculture and the elimination of important grazers, such as bison (*Bison bison*), which kept the habitat sparsely vegetated, began in the 1800s (McGaugh 2006). Currently, loss of traditional wintering sites on grasslands and suitable agricultural cropland to urban development, vineyards, or other incompatible land uses could continue to reduce suitable wintering habitat for mountain plover (Hunting and Edson 2008). In addition to allowing higher vegetation structure that is unsuitable for mountain plover, incompatible agricultural practices can directly kill plovers from farm equipment or expose plovers to pesticides (McGaugh 2006). Grain fields, which have become more popular in the last 25 years, remain fallow until early May, after most mountain plovers have started nesting, and farm equipment destroys many nests when fields are planted in May (Knopf and Wunder 2006). However, plovers will successfully renest on tilled fields, and although the transition to more grain crops was originally thought to have a substantial impact on mountain plover reproduction, this factor has since been dismissed as having an additive impact beyond normal nest-predation rates (Knopf and Wunder 2006). High levels of organochlorine residues were found in birds collected from California's Imperial, San Luis Obispo, and Tulare Counties in 1991–1992 (Knopf and Wunder 2006). However, there is no evidence that mountain plover reproductive success or survival is affected by pesticide use (McGaugh 2006).

Predation is the main source of egg and chick loss. Mountain plovers are susceptible to a variety of predators, such as birds, mammals, and reptiles (County of Riverside 2003; McGaugh 2006). Reduced populations of fossorial mammals could impact mountain plover populations since they attract invertebrates used for forage (Hunting and Edson 2008).

Mountain plover is also susceptible to extreme weather conditions. At the Pawnee National Grassland in Colorado, hail and flooding caused almost complete reproductive failure (McGaugh 2006). Climatic conditions also influence vegetation structure with wetter years possibly supporting fewer wintering mountain plover (76 FR 27756–27799).

Because mountain plovers tend to be unwary and form tight flocks, they have historically been susceptible to hunters (e.g., in the late 1800s) (McGaugh 2006; Knopf and Wunder 2006). However, shootings in more recent years have not been documented, and hunting is not a current conservation concern (Knopf and Wunder 2006). Although very tolerant of machinery, such as off-road vehicles, tractors, and military aircraft, mountain plovers will flee nest sites or roost areas when approached by humans on foot, leaving eggs susceptible to overheating due to solar radiation (Knopf and Wunder 2006).

Conservation and Management Activities

Mountain plover is not the subject of a documented recovery plan, and there do not appear to be any active state or local programs focused on its conservation and management. However, it is a Covered Species in several approved habitat conservation plans and natural community conservation plans. Several of these are not related to the Plan Area, but the mountain plover is also a Covered Species under the Bureau of Land Management's (BLM's) West Mojave Plan, which proposes possible management actions such as subsidizing alfalfa farmers, establishing agricultural preserves, and encouraging land-use practices that benefit mountain plovers. These might include periodically disking and/or burning fields or controlling the use of pesticides (McGaugh 2006).

Data Characterization

Both breeding and wintering mountain plover populations should be monitored more carefully. Monitoring of wintering populations in California should focus on traditional wintering sites and high-quality habitat in the Imperial Valley, Carrizo Plain, Panoche Valley, and Central Valley. Standardized methods should be used that sample in order to estimate a statewide wintering population if possible. Other data collected should include land-use changes, habitat suitability, and annual habitat availability. Mountain plover life history and

distributional characteristics should be considered to enable the development of population estimates at 3- to 5-year intervals (Hunting and Edson 2008).

Additional research could focus on determining the potential effects of mountain plover chronic exposure to agrochemicals in the Central and Imperial Valleys and to determine whether there is a relationship between agrochemical use and winter plover distribution in response to changes in prey selection, availability, and abundance (Hunting and Edson 2008).

Research can also focus on documenting the differential seasonal use by plovers of native and non-native grasslands as opposed to cultivated lands and determine what factors drive the shifts between habitats. The correlations between the availability and suitability of habitat and winter survival, movement patterns, and foraging strategies can also be explored (Hunting and Edson 2008).

Management and Monitoring Considerations

Management should focus on protecting traditional wintering sites and high-quality wintering habitat from urban development and conversion to other incompatible land uses. This can be achieved by securing conservation easements and property acquisition as part of regional conservation planning efforts (Hunting and Edson 2008). Furthermore, the subsidization of alfalfa farmers, establishment of agricultural preserves, and encouragement of land-use practices that benefit mountain plovers can also help preserve suitable habitat for this species (McGaugh 2006).

Habitat quality for mountain plover can be maintained by management of grasslands at low stature and density (Hunting and Edson 2008). Periodic disking and/or burning fields or controlling the use of pesticides can also maintain habitat for the species (McGaugh 2006).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for mountain plover, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled

suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 718,451 acres of modeled suitable habitat for mountain plover in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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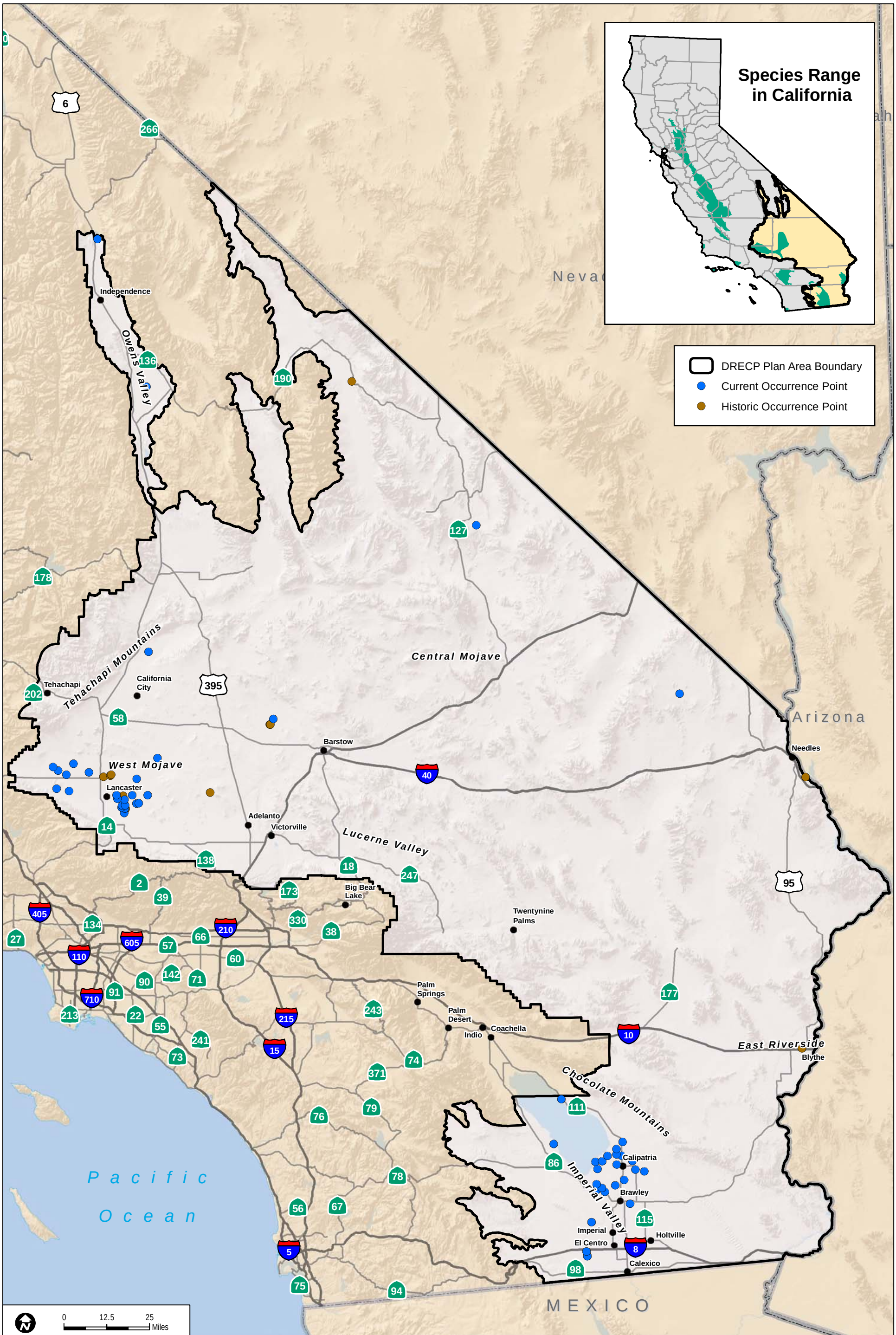

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B09

Mountain Plover Occurrences in the Plan Area

Swainson's Hawk (*Buteo swainsoni*)

Photo by Dudek.

Legal Status

State: Threatened
Federal: U.S. Fish and Wildlife Service Bird of Conservation Concern
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

The Swainson's hawk (*Buteo swainsoni*) is monotypic with no currently accepted subspecies (Bechard et al. 2010). It is most closely related to the Galapagos hawk (*B. galapagoensis*) (Bollmer et al. 2006, Hull et al. 2008), which, combined with their migration patterns, indicates a South American origin for this species (Mayr and Short 1970). Hull et al. (2007) examined the genetic diversity of Swainson's hawks throughout their North American breeding range and concluded that California's Central Valley population was genetically distinct from other populations, although the distinction was not great enough to meet the standards for an evolutionarily significant unit, as defined by Moritz (1994) as a historically isolated set of populations. Moritz (1994) further stated that "ESUs should be reciprocally monophyletic for mtDNA and alleles and show significant divergence of allele frequencies at nuclear loci." Limited dispersal data suggest that populations from different parts of the breeding range do not readily mix on their South American wintering grounds (Woodbridge et al. 1995a). Further, the California Central Valley hawks have recently established a wintering population in southwestern Mexico and a small wintering population of about 30 birds in Sacramento-San Joaquin River Delta in the Central Valley (Herzog 1996; Wheeler 2003; Bradbury unpublished data). These observations support the hypothesis that Swainson's hawks from California's Central Valley are distinct from populations elsewhere including birds from the southwestern deserts which are most closely related to birds from the

Great Basin and Great Plains. Work conducted by Woodbridge in Butte Valley in northeastern California found that the behavior of the Central Valley population was different from the Butte Valley population (England pers. comm. 2012).

Distribution

General

Swainson's hawks breed in the grasslands, shrub-steppe, desert, and agricultural areas of the Columbia Basin, Great Basin, Great Plains, American Southwest, and the Central Valley of California (Bechard et al. 2010) (Figure SP-B12). In California, approximately 94% of the breeding pairs now occur in the Central Valley (CDFG 2007) with most found between Modesto and Sacramento (Bloom 1980). Smaller California breeding populations are also found in the Great Basin in the extreme northeastern California portion of the state, in the Owens River Valley, and in nearby Fish Lake Valley on the Nevada border. Remnant (or recolonizing) populations in Southern California are found in the western Mojave Desert in the Antelope Valley and in the eastern Mohave Desert in the Mojave National Preserve. Historically, Swainson's hawks nested throughout the California lowlands, including coastal valleys and plains where they no longer occur today (Bloom 1980).

Specific locations where Swainson's hawks have been reported breeding in southeastern California include near Cima Dome and Lanfair Valley in San Bernardino County, at Oasis Ranch in Mono County, and near Lancaster in Los Angeles County. The species formerly bred in Joshua tree woodland habitat near Victorville and Adelanto in San Bernardino County (England and Laudenslayer, cited in Latting and Rowlings 1995).

Migrating Swainson's hawks pass through Anza Borrego State Park and Morongo Valley in spring. In fall, hawks also migrate through the eastern Colorado Desert and along the Colorado River. While most birds winter in South America, there are small, isolated wintering populations in the Sacramento-San Joaquin River Delta in California and in southern Florida (Natural Resource Consultants and Western EcoSystems Technology, Inc. 2011), as well as Mexico (England, pers. comm. 2012).

Distribution and Occurrence within the Plan Area

Historical

Historically, Swainson's hawks were much more common in the Southern California deserts than they are today (Sharp 1902; Bloom 1980). Bloom (1980) estimated that the Mojave/Colorado Deserts population declined by 95% in the previous century. Current nesting territories in Southern California may represent recolonizations (Woodbridge 1998). There are four historical (i.e., pre-1990) occurrence records in the Plan Area and an additional three records with an unknown observation date (CDFW 2013; Dudek 2013). The four historical occurrences with known observation dates include a 1927 occurrence east of Lancaster and south of E. K8, and 1979 and 1982 occurrences in the eastern portion of the Mojave National Preserve (Figure SP-B12). The latter three historical nest territories in the Lanfair Valley within the Mojave National Preserve had last reported activity in the early 1980s. The occurrences with no observation date in the Dudek (2013) dataset include a site along E. Avenue I east of Lancaster, a site along E. Avenue J east of Lancaster (both of which are north of the 1997 occurrence east of Lancaster), and site north of Fremont Wash and east of State Highway 395 (Figure SP-B12).

Recent

There are 52 recent (i.e., since 1990) occurrences for Swainson's hawk in the Plan Area (CDFW 2013; Dudek 2013) (Figure SP-B12). Most breeding pairs within the DRECP area are located in the western Mojave along the base of the San Gabriel and Tehachapi Mountains and in the Antelope Valley. Approximately ten pairs nest over a relatively wide area in the Antelope Valley (Bloom 2011). Several pairs nest in the upper Owens River Valley, just north (outside) of the DRECP area. However, an isolated Owens River Valley nesting territory (active in 2003) does occur inside the DRECP area at Haiwee Reservoir (Bloom 2011). Scattered recent occurrences are located in the Fremont Valley, the Ridgecrest/China Lake Naval Air Weapons Station, and near Haiwee Reservoir. There is a single occurrence south of the Salton Sea from 2003.

Natural History

Habitat Requirements

Swainson's hawks are primarily a grassland bird but they are also found in sparse shrubland and small, open woodlands (Bechard et al. 2010). In Central California Swainson's hawks are primarily associated with grain and hay croplands that mimic native grasslands with respect to prey density and availability (Estep 1989; Babcock 1995). They generally nest in isolated trees, narrow bands of vegetation, or along riparian corridors in grassland, shrubland, and agricultural landscapes. Within the DRECP area, Joshua trees (*Yucca brevifolia*) and non-native ornamental trees or trees planted as windbreaks also function as nest sites (CEC and CDFG 2010; Table 1).

Most Swainson's hawks winter in the pampas (grasslands) of South America, but there they have adapted to agricultural lands, as they have on their North American breeding grounds (Woodbridge et al. 1995a). Foraging habitat includes dry land and irrigated pasture, alfalfa, fallow fields, low-growing row or field crops, new orchards, and cereal grain crops. In the Plan Area, in addition to alfalfa fields in the Antelope Valley, Swainson's hawks may also forage in grasslands, Joshua tree woodlands, and other desert scrub habitats that support a suitable prey base.

Table 1. Habitat Associations for Swainson's Hawks in the Plan Area

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Cropland	Foraging; nesting	Primary	Adapted to foraging in agricultural fields, but not in crops that grow higher than native vegetation. Nests in isolated trees or in adjacent riparian vegetation	Direct observations
Joshua tree woodlands	Nesting	Secondary	Historically nested in Joshua tree woodlands, now also in ornamental roadside trees and	Direct observations

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
			wind row trees (see above)	
Desert grasslands	Foraging	Primary	Forages in open landscapes with low and/or widely spaced vegetation	Direct observations
Desert scrub	Foraging	Secondary	See above	Direct observations

Sources: Bechard 1982; CEC and CDFG 2010; Estep 1989

Foraging Requirements

In North America, breeding Swainson's hawks prey chiefly upon small rodents such as young ground squirrels (*Spermophilis* spp.), pocket gophers (*Thomomys* spp.), deer mice (*Peromyscus* spp.), and voles (*Microtus* spp.). Voles are especially important to Central California hawks. Their breeding season diet also includes birds, snakes, and insects (especially grasshoppers and crickets) (Snyder and Wiley 1976; Fitzner 1980; Bednarz 1988; Estep 1989). Non-breeding birds in North America and wintering birds in South America feed almost exclusively on insects, especially grasshoppers (Synder and Wiley 1976; Johnson et al. 1987; Sarasola and Negro 2005).

In addition to insects, Swainson's hawks in the Antelope Valley forage primarily on Botta's pocket gopher *(Thomomys bottae*) in agricultural areas and on a wider variety of prey in desert scrub and grassland habitats (CEC and CDFG 2010).

Reproduction

Swainson's hawks arrive on the breeding grounds in March-April (March in Central California) (Table 2) and begin a week-long nest building phase 1 to 2 weeks after arrival (Fitzner 1980). The egg-laying through fledging period lasts about 73 days per nest, but can last 110 days for the local population (Olendorff 1973). Adjacent pairs can be out of sync by 25 days (Woodbridge 1987). Typical clutch size is 2 or 3 eggs (Olendorff 1973; Fitzner 1980; Bechard 1983; Bednarz

and Hoffman 1986) and typically about 2 young are fledged per successful nest (range of 1.62 to 2.18) (Bechard et al. 2010. A study of rural and urban nest sites central California found 1.65 and 1.64 young fledged per successful nest site, respectively (England et al. 1995). The number of fledglings can average less than 1 during years of low prey availability (i.e., not all nests are successful) (Bechard 1983). Young generally fledge mid-July to mid-August at an average age of 43 days (Olendorff 1973, Fitzner 1980, Woodbridge 1987).

Table 2. Key Seasonal Periods for Swainson's Hawks

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding				✓	✓	✓	✓	✓				
Migration		✓	✓						✓	✓		
Wintering	✓	✓									✓	✓

Notes: Central Valley (California) Swainson's hawks arrive a month earlier on breeding grounds than other populations, possibly because they winter in central Mexico (Bradbury unpublished) rather than Argentina.

Sources: Wheeler 2003, Bechard et al. 2010

Spatial Behavior

Spatial behaviors by Swainson's hawk include migration patterns, breeding home range use, and natal dispersal.

Migratory movements occur annually between North American breeding grounds and wintering areas primarily located in South America, although some Swainson's hawks use wintering grounds in California and Mexico (Fuller et al. 1998; Bechard et al. 2010; Wheeler 2003; Bradbury unpublished data). Immature birds and post-breeding adults begin forming migration flocks in August and September, and begin the fall migration in September. Birds migrating to South America leave North America by October and arrive in Argentina in November (Bechard et al. 2010). The return migration begins late-February and early March in Argentina (Bechard et al. 2010), with birds arriving in California from early March (Central

Valley) through April (other California populations). Fuller et al. (1998) tracked 27 Swainson's hawks on their 1996 and 1997 southbound migrations and recorded a mean cumulative travel distance of over 13,500 kilometers (8,370 miles).

Local movements of California hawks are primarily confined to home ranges, which vary greatly in size (from 69 to 8,718 hectares) among populations (Bechard et al. 2010). Smaller home ranges (e.g., less than 1,000 hectares) tend to occur areas with suitable foraging habitat such as alfalfa, fallow fields and dry pastures, while large home ranges (e.g., greater than 2,500 hectares) tend to occur in areas less suitable foraging habitat, such as mature grains and row crops, vineyards, and orchards (Bechard et al. 2010). Natal dispersal also varies greatly among populations. Central California hawks disperse only a few kilometers (mean of 3.5 kilometers; Estep 1989), while northeastern California hawks disperse farther (mean of 9 kilometers) (Woodbridge et al. 1995b). But in greater contrast, juvenile Swainson's hawks in Saskatchewan apparently disperse to distances exceeding 200 kilometers (Houston and Schmutz 1995).

Table 3. Movement Distances for Swainson's Hawks

Type	Distance/Area	Location of Study	Citation
Home Range	69–8,718 ha	Washington, Oregon	Fitzner 1978; Bechard; 1989; Woodbridge 1991
Dispersal Range	3.5–9 km	California	Estep 1989; Woodbridge et al. 1995b
Migration	Mean of 13,504 km southward, 11,592 km northward	United States	Fuller et al. 1998

Notes: ha = hectare; km = kilometer. Home range depends on habitat type.

Ecological Relationships

Predator–prey relationships are critical for Swainson's hawk. Conversion of suitable nesting and foraging habitat in some locations in North America, and especially Central California (Risebrough et al. 1989), has led to the loss of nesting opportunities and reduction of

prey populations due to conversion of native grassland to cropland. Where agricultural conversion has been to crop types not suitable for foraging and alternative nesting opportunities have not been created, Swainson's hawk populations have dexlined (Bloom 1980; Bechard et al. 2010). Also, because of their dependence on insect prey, especially grasshoppers on the wintering grounds, Swainson's hawks are highly susceptible to secondary poisoning from insecticides (Woodbridge et al. 1995a).

Swainson's hawks occasionally lose nestlings or fledglings to great horned owl (*Bubo virginianus*) predation (Fitzner 1978; Littlefield et al. 1984; Woodbridge 1991), and Swainson's hawks themselves have preyed on burrowing owl (*Athene cunicularia*) fledglings (Clayton and Schmutz 1999). Interspecific competition and territoriality occurs between Swainson's hawk and sympatric buteos (e.g., red-tailed hawks [*Buteo jamaicensis*]) over control of nest sites, although Swainson's hawks appear to dominate in most such encounters (Janes 1984).

Population Status and Trends

Global: Secure (NatureServe 2010)
State: Imperiled (NatureServe 2010)
Within Plan Area: Imperiled (CEC and CDFG 2010)

In California, Swainson's hawk is vulnerable to extirpation due to its very restricted range (primarily the Central Valley), few populations, steep population declines, and loss of habitat. Bloom (1980) concluded that the California Swainson's hawk population had declined 90% since 1900 when Sharp (1902) considered the species abundant. Much of this decline occurred in Southern California, where the species was once considered abundant in coastal valleys (Sharp 1902) but is now completely absent. Based on its large decline, Swainson's hawk was listed as a state-threatened species in 1983. Later inventories estimated populations of 800 hawks in 1988 and 1,000 hawks in 1994 (CDFG 2007). The CDFG initiated an inventory of Swainson's hawk breeding pairs in California in 2005 and 2006 (CDFG 2007a). Based on a randomized sampling, the CDFG estimated a breeding population of 1,912 pairs (95% confidence interval of 1,471 to 2,353 pairs) in 2005 and 2,251 breeding pairs (95% confidence interval of 1,811 to 2,690 pairs) in 2006. The combined estimate for

2005–2006 is 2,081 pairs (95% confidence interval of 1,770 to 2,393 pairs). Approximately 94% of the breeding pairs now occur in the Central Valley.

Swainson's hawk populations in the Mojave and Colorado desert portions of the DRECP area have also declined severely in the past century. Bloom (1980) estimated that this region once supported 270–1,080 pairs, but abundance has since declined as much as 95%. Today, a few nesting pairs occur in Antelope Valley at the extreme western edge of the Mojave Desert and primarily forage in the alfalfa fields and other agricultural areas in the region (CEC and CDFG 2010; Bloom 2011). They also forage in grassland, Joshua tree woodlands, desert scrub habitats (CEC and CDFG 2010). A small breeding population has been identified at Mojave National Preserve near the Nevada border (CNDDB 2011). The Owens Valley population is principally found immediately north of the DRECP boundary, but there is one record inside the Plan Area south of Owens Lake, and in the future the Owens Lake population may further expand into the Plan Area. These small, isolated populations could be remnants of the much larger historical population, or they could be recent colonists, in which case the Southern California population would be growing.

Threats and Environmental Stressors

The decline of Swainson's hawks in California has been attributed to riparian habitat loss and agricultural and urban development in the Central Valley (Bloom 1980; England et al. 1995), urbanization in the coastal valleys and plains (Bloom 1980), and a contracting range of Joshua trees and riparian habitats in the Mojave Desert (Bloom 1980). It was estimated that by the mid-1980s, approximately 93% of riparian habitat in the San Joaquin Valley and 73% of riparian habitat in the Sacramento Valley had been lost since the 1850s (CDFG 1994). Chronic and acute pesticide poisoning also affects the Swainson's hawk (Goldstein et al. 1996; Risebrough et al. 1989). Pesticide use on South American wintering grounds threatens all North American populations. South American birds have died from ingesting pesticides targeting grasshoppers (Woodbridge et al 1995a; Goldstein et al. 1996). Goldstein et al. (1996) estimated that 4,100 Swainson's hawks died in 1 year, 1996, from acute pesticide poisoning in Argentina.

Wildfires, lowering of water tables, and flood control also continue to threaten riparian and woodland nesting habitat in California. Off-road vehicle activity and shooting can also disrupt nesting, although the latter is not as important a factor as it once was. Intraspecific competition or aggression with other raptors and common ravens (*Corvus corax*) has been suggested as a stressor elsewhere in the western United States (Janes 1987; Littlefield et al. 1984).

Conservation and Management Activities

There are no active conservation efforts specific to Swainson's hawks in the DRECP area. The CEC and CDFG have developed protocols to avoid and minimize impacts of renewable energy projects on Swainson's hawk in the Antelope Valley (CEC and CDFG 2010). These protocols include methods for conducting pre-project surveys within a 5-mile radius of a proposed project. If active nests are found in proximity to a project a Monitoring and Mitigation Plan is required. Potential avoidance and minimization measures include maintaining sufficient foraging and fledgling area; providing a 0.5-mile buffer zone during construction between project activities and an active nest; avoiding nest trees to extent feasible; and providing habitat management lands to offset habitat losses within 0.5 mile of an active nest. The overarching objective of these protocols avoid significant impacts to nesting and foraging individuals and thus to enable renewable energy projects to comply with CEQA and CESA regulations regarding the Swainson's hawk.

Further, the Los Angeles Audubon Society is focusing conservation efforts towards the approximately ten pairs of Swainson's hawks inhabiting the Antelope Valley. This effort has been largely confined to encouraging the City of Lancaster to consider Swainson's hawk conservation in any future solar energy permitting.

The Desert Bird Conservation Plan, jointly developed by the California Partners in Flight (CalPIF) and Point Reyes Bird Observatory (PRBO) Conservation Science, is a non-regulatory document designed to assist land-managers in improving habitat condition for desert birds of the Mojave and Colorado Deserts (the portion of the Sonoran Desert in the Plan Area). Although Swainson's hawks are not a focal species in the Desert Bird Conservation Plan, the plan does promote restoration

of Joshua tree habitats that are important to nesting Swainson's hawks. Statewide, Swainson's hawks are a focus of the CalPIF/PRBO Riparian Bird Conservation Plan, which recognizes the importance of riparian trees (e.g., Fremont cottonwood [*Populus fremontii*]) as nesting habitat for California Swainson's hawks.

The Friends of the Swainson's Hawk, a grassroots organization founded in 1994, recently developed a conservation strategy for California Swainson's hawk populations. Although this strategy focuses on Central Valley populations, it does provide a framework for conservation and management of Swainson's hawks statewide.

The CDFG also published a staff report in 1994 regarding recommended mitigation for Swainson's hawk that includes recommendations for mitigation for impacts within a 10-mile radius of an active nest site; the 10-mile radius reflects common flight distances between an active nest and foraging habitat (CDFG 1994).

Data Characterization

The current status of nesting territories in the Owens River Valley and the Mojave National Preserve within the DRECP area is unknown. It is likely, however, that most of the Swainson's hawk concerns relative to DRECP will be in the western Mojave region where the large majority or nesting sites occur.

Management and Monitoring Considerations

Within the DRECP area, management and monitoring considerations include maintaining suitable nesting habitat and proximity to reliable food sources. Currently Swainson's hawks rely heavily on the alfalfa and other agricultural fields for prey (primarily gophers and insects), but they may also forage in desert scrub and Joshua tree woodland habitats within flight distances from active nests (CEC and CDFG 2010; Bloom 2011). Potential disturbance of active nest sites from human activities is also a concern.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Swainson's hawk, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 1,615,796 acres of modeled suitable habitat for Swainson's hawk in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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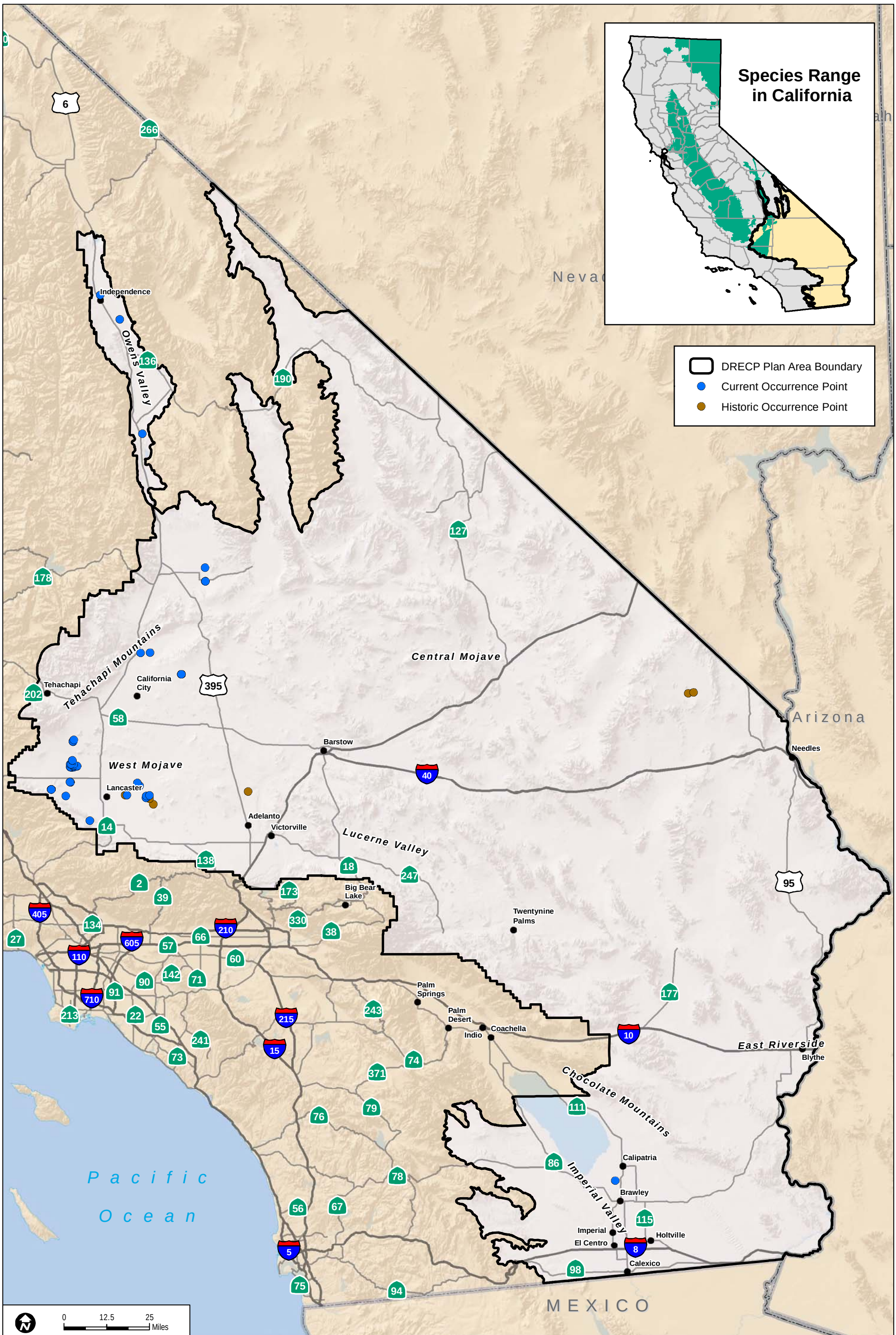

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B10

Swainson's Hawk Occurrences in the Plan Area

Burro Deer (*Odocoileus hemionus eremicus*)

Legal Status

State: None
Federal: None
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

The burro deer (*Odocoileus hemionus eremicus*) is the desert dwelling subspecies of the widespread mule deer (*Odocoileus hemionus*). The burro deer was first described by Mearns in 1897 from a specimen taken near the Gulf of California in Sonora, Mexico. Longhurst and Chatting (as cited in Celentano and Garcia 1984) reported that burro deer are distinguished from other subspecies on the basis of cranial measurements, external body measurements, and coloration. Since 1997, desert mule deer (*O. h. crooki*) and burro deer (*O. h. eremicus*) have been synonymized (*O. h. eremicus*) (Heffelfinger 2006). As a result, the overall area identified as containing this subspecies now encompasses much of the southwestern United States and northern Mexico, including southeastern California (Marshal et al. 2004).

Distribution

General

Mule deer are widespread across most of the western United States, western Canada, and south into northern Mexico. The burro deer subspecies is native to the Mojave and Sonoran deserts of the southwestern United States and northern Mexico. Within California, the burro deer is found in the eastern portions of Imperial and Riverside counties, and as far north as the southeastern corner of San Bernardino County. From the Colorado River they range west into California along vegetated washes to the Coxcomb Mountains, Palen Mountains, Little San Bernardino Mountains, Chuckwalla Mountains,

Chocolate Mountains, and formerly through the Imperial Valley to Indio. Burro deer are predominately associated with major river corridors and dry desert washes leading down to the Colorado River and other major rivers. In the hottest months deer are found close to permanent water and forage sources such as the Colorado River. However, with the onset of the summer monsoons in early August and September, burro deer may disperse to the desert mountains (Celentano and Garcia 1984).

Distribution and Occurrences within the Plan Area

Historical

The distribution of burro deer within California was described as far back as 1936 and appears to reflect their current distribution, though it is thought that their former range extended northwest through the Imperial Valley to Indio, and may once have extended around the west side of the Salton Sea (Celentano and Garcia 1984). Much of the area west of Salton Sea and north to Indio was converted to agriculture several decades ago. No pre-1990 occurrences are recorded within the California Natural Diversity Database (CNDDB); however, annual harvest population estimates indicate that the burro deer population fluctuated between 2,000 and 5,000 individuals between 1940 and 1990 (Celentano and Garcia 1984; CDFG 1997, 2007).

Recent

There is no evidence to suggest that burro deer distribution differs from historical (pre-1990) distribution described above. Because burro deer is not a state special-status species, it is not tracked in the CNDDB. However, data compiled by the Conservation Biology Institute (CBI) includes at least six mapped occurrence locations within the Desert Renewable Energy Conservation Plan (DRECP) Area (Figure SP-M02) (Data Basin 2013). Three of the occurrences were along or near the Colorado River, including one near Blythe and the other two in the Palo Verde Area. Two adjacent occurrences are located in the Smoketree Valley area and the other occurrence is near Clemens Well in the valley between the Orocopia and Chocolate mountains. The most recent available estimates made to assist with

hunting and herd management put the current burro deer population at about 2,000 individuals (CDFG 2007).

Natural History

Habitat Requirements

The burro deer is a large ungulate that shifts seasonally between desert riparian washes and more open, mountainous terrain. It depends on the availability of water and tracks the best available forage throughout the year. Burro deer need to drink at least every 3–4 days, but tend to drink each night, and therefore require predictable water sources. Consequently, their seasonal distribution is closely associated with water availability (Celentano and Garcia 1984).

During the driest season, between January and March, deer concentrate in lowland riparian habitats, including riparian forest, alluvial and riparian scrub, and alluvial woodland, where water is predictable and forage vegetation quality is relatively high. With the onset of the summer monsoonal rains in July and August, burro deer are less constrained by water sources and use the network of alluvial and wash communities to migrate between lowland riparian communities and the mountainous desert communities that include Sonoran Desert scrub, alluvial woodland, and Joshua tree woodland (Celentano and Garcia 1984; Marshal et al. 2006a) (Table 1). Burro deer remain at high elevations throughout the autumn and winter (Marshal et al. 2006a), only returning to more predictable forage and water sources at lower elevations in spring (Table1).

Burro deer track the highest quality forage, which depends on monsoonal and winter rainfall. Monsoonal rainfall in particular can be highly localized, and consequently forage quality is very heterogeneous (Marshal et al. 2006a, 2006b). As a result, burro deer abundance and distribution can be highly variable from year to year (Marshal et al. 2006c).

Table 1. Habitat Associations for Burro Deer

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Riparian Forest; Alluvial and Riparian Scrub; Alluvial Woodland; Desert Dunes.	Shelter and foraging	Spring, early Summer	Xeroriparian washes, riparian habitats used for shelter and foraging.	Celentano and Garcia 1984; Marshal et al. 2006a
Sonoran Desert Scrub; Alluvial Woodland; Joshua Tree Woodland.	Rutting/ fawning/ foraging	Summer/ Autumn/ Winter	Females and fawns steeper slopes, avoiding ridges and valley flats.	Marshal et al. 2006a; Marshal et al. 2006c

Foraging Requirements

Burro deer foraging patterns vary seasonally and are dictated by water availability and quality of forage plants (Marshal et al. 2006a). Their forage is dominated by browse and forbs, with only 10% of their diet consisting of grasses and succulents (Krausman et al. 1997; Marshal et al. 2006b, 2012). During the driest season, in spring and pre-monsoonal summer, burro deer are closely associated with water sources and, consequently, rely on riparian, xeroriparian, and desert wash communities that produce most of the high-quality forage. Forage plants include catclaw (*Acacia greggii*), desert ironwood (*Olneya tesota*), palo verde (*Parkinsonia florida*), honey mesquite (*Prosopis glandulosa*), and cheese bush (*Hymenoclea salsola*). Deer foraging adjacent to the Colorado River include salt cedar (*Tamarix* spp.), cattails (*Typha domingensis*), and arrowweed (*Pluchea sericea*) in their diet (Marshal et al. 2004, 2006b, 2012).

Following the onset of the monsoon between late July and early August, burro deer are less constrained by water sources and are found on steeper ground at high elevations (Marshal et al. 2006a).

Common forage plants for burro deer in piedmont and mountainous areas are creosote bush (*Larrea tridentata*), burro-weed (*Ambrosia dumosa*), brittle-bush (*Encelia farinosa*), and ocotillo (*Fouquieria splendens*) (Marshal et al. 2006b).

As noted above, burro deer forage is dominated by browse vegetation. Microhistological examination of deer pellets found that diets of burro deer had high proportions of browse (76%–85%) in all seasons and low proportions of grasses (1%–2%) and forbs (4%–8%). Browse plants were dominated by saltbush (*Atriplex* spp.), Mexican tea (*Ephedra californica*), desert ironwood, palo verde, and honey mesquite (Marshal et al. 2004, 2012).

Reproduction

Burro deer tend to rut and mate later than most mule deer (Heffelfinger 2006). Rutting and mating may occur as early as late December and as late as March (Table 2) (Celentano and Garcia 1984; Marshal et al. 2006a).

Fawning occurs between July and mid-October (Table 2), timed to take advantage of summer monsoon rains. Fawning occurs in both riparian and mountainous desert habitats, although observations made during fawning indicate that it occurs in areas characterized by low hills with a network of interconnecting washes (Celentano and Garcia 1984). Does with fawns then move into more mountainous terrain where they have a tendency to avoid valley floors and ridges, which are associated with higher predator densities (Marshal et al. 2006a). Fawns are believed to be susceptible to coyote (*Canis latrans*) and golden eagle (*Aquila chrysaetos*) predation until they are at least 6 months old (Marshal et al. 2006a).

Table 2. Key Seasonal Periods for Burro Deer

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Rutting/ Breeding	X	X	X									
Migration							X	X				
Fawning/ rearing of young							X	X	X	X	X	

Sources: Celentano and Garcia 1984; Marshal et al. 2006a

Spatial Activity

Burro deer generally follow a seasonal migratory pattern in the Plan Area. During the drier spring and summer periods, burro deer occur in riparian woodlands and washes bordering major water sources such as the Colorado River, Coachella Canal, or All American Canal. As the summer monsoonal rains arrive, between late July and August, burro deer migrate to the desert mountains, coinciding with the flush of new growth for desert forage plants and raising fawns (Celentano and Garcia 1984). Burro deer only shift back to the lowlands in spring as temporary waters sources dry out. Migration is not universal, however, and some burro deer remain around permanent water sources in the Chocolate Mountains (Celentano and Garcia 1984).

Home range patterns vary considerably between seasons. During the hot spring and summer months, deer are restricted to permanent water sources and do not range far. Burro deer occupying Colorado River riparian woodlands may have home range as small as 1 square mile, while deer in dry wash woodland may have home ranges of 2–8 square miles (Celentano and Garcia 1984). During the cooler winter months, when movement is not restricted by water or high temperatures, individual ranges in the mountains may cover 30–50 square miles (Table 3).

Table 3. Movement Distances for Burro Deer

Type	Distance/Area	Location of Study	Citation
Home Range Summer	1–8 square miles		Celentano and Garcia, 1984
Home Range Winter	15–30 square miles		Celentano and Garcia 1984

Ecological Relationships

Rainfall has an important influence on mule deer populations in the deserts of Southern California, with both abundance and population dynamics related to the amount of rainfall. Forage resources in deserts are affected primarily by rainfall, which is highly variable seasonally between years and across the range. As a result, resource availability and its influence on deer populations is highly variable from year to year (Marshal et al. 2002, 2005). Despite these general relationships, however, there is currently no direct evidence linking burro deer population dynamics to the large-scale climatic variation caused by El Niño southern oscillation events (Marshal and Bleich 2011).

During the summer monsoonal season, rainfall events tend to produce strip rains, where a large amount of rain falls on an area about 1 kilometer wide and several kilometers long, with little rain falling on adjacent areas. Strip rains produce a highly heterogeneous response in plant growth (Marshal et al. 2005) and a patchy distribution of forage biomass and quality. Burro deer respond to this heterogeneity by selecting areas with rapidly growing plants, such as those in areas that recently received rainfall, because forage from those plants are high in water, protein, and digestibility. When rapidly growing forage is not available, deer may select areas of high forage biomass, where they can take advantage of forage of higher digestibility before plant biomass and digestibility decrease. When forage water decreases beyond a critical threshold, however, locations of permanent water, including catchments, may become most important in determining deer distribution, and forage growth and biomass become secondary to water availability (Marshal et al. 2005).

It is unclear to what degree mule deer compete or interact with other large- and medium-sized herbivores in the area, such as bighorn

sheep (*Ovis canadensis*), feral ass (*Equus asinus*), black-tailed jackrabbit (*Lepus californicus*), desert cottontail (*Sylvilagus audubonii*), and desert tortoise (*Gopherus agassizii*). Studies assessing the overlap between deer and the feral ass indicate biologically significant overlap, but with the burro deer diet containing more browse and forbs and significantly less grass than the ass (Marshal et al. 2012). Burro deer and bighorn sheep may share diets where their habitats overlap, but they exhibit seasonal separation. In the driest periods of spring and summer, when bighorn sheep may use desert washes, burro deer tend to concentrate in riparian habitats.

Potential predators of burro deer include mountain lion (*Puma concolor*), coyote, bobcat (*Lynx rufus*), and golden eagle. However, the extent to which predators affect burro deer populations is currently unknown. Marshal et al. (2006a) suggest that predators, particularly coyote, may be responsible for females with fawns avoiding valley floors and ridges until the fawns are at least 6 months old. Predator exclusion experiments in Arizona have shown that predation is a significant factor in fawn mortality (Heffelfinger 2006).

Population Status and Trends

Global: Secure (NatureServe 2012)
State: Stable
Within Plan Area: Stable

Burro deer are not currently listed as threatened or special status, but are managed in California for their recreational, educational, and hunting value. Available evidence suggests the population is stable. Past surveys estimated a population of about 2,000 individuals (Celentano and Garcia 1984), with estimates in the 1980s and 1990s varying between 2,000 and 5,000 individuals (CDFG 1997). More recent estimates in the early 2000s from telemetric and remote photographic studies estimate herd densities of 0.05–0.13 deer per square kilometer (Marshal et al. 2006c), indicating a population in the in the range of 970 and 2,500 individuals.

For hunting purposes, population trends and herd health have generally been inferred from harvest data, climatic conditions, and plant productivity (Celentano and Garcia 1984). However, deer harvests

observed a fourfold increase between 1948 and 1998 (Marshal et al. 2002). Such an increase is a reflection of increased hunting intensity and changes in reporting methods for harvested deer (Celentano and Garcia 1984; CDFG 1997). The increased hunting intensity has, thus far, had no detectable effect upon the population. Current population size and composition are estimated from harvest models, developed in the mid-2000s. The most recent available estimate for 2007 puts the population close to historical levels: 1,940 individuals in 2007 compared to 2,000 individuals in 1940 (CDFG 2007).

Estimates of herd composition are highly variable (Table 4). Celentano and Garcia (1984) estimated sex and age ratio using aerial and ground telemetry, and Thompson and Bleich (1993) tested the efficacy of ground, aerial, and hunter surveys in estimating herd composition but did not estimate abundance. The most recent population estimates for the East Chocolate–Cargo Muchacho area concluded that burro deer occur at densities between 0.05-0.13 deer per square kilometer. This estimate is comparable to the historical estimates of deer densities of 0.08 deer per square kilometer in 1940 and 0.11 deer per square kilometer in 1952 (Marshal et al. 2006c).

The extensive telemetry and remote photography studies conducted between 1999 and 2004 focused on demographic composition, habitat utilization, and potential interactions with other large herbivores such as feral ass. It is evident from these most recent studies that observed abundance and density are highly variable between years, and consequently estimating long-term trends in herd size and health from just a few years of data is difficult (Marshal et al. 2006a, 2006b, 2006c, 2012; Marshal and Bleich 2011).

Table 4. Estimated Herd Composition Ratios from Three Studies of Burro Deer in California

Year	Female	Young	Male	Method
1981[1]	100	65	No estimate	Aerial and ground telemetry
1982[1]	100	56	No estimate	Aerial and ground telemetry
1990[2]	100	25	35	Aerial survey
	100	43	29	Ground survey
	100	35	31	Hunter interviews
1999[3]	100	28	9	Remote photography and aerial telemetry
2000[3]	100	17	33	Remote photography and aerial telemetry
2001[3]	100	10	55	Remote photography and aerial telemetry
2002[3]	100	71	38	Remote photography and aerial telemetry
2003[3]	100	43	40	Remote photography and aerial telemetry
2004[3]	100	85	61	Remote photography and aerial telemetry

[1] Celentano and Garcia 1984
[2] Thompson and Bleich 1993
[3] Marshal et al. 2006c

Threats and Environmental Stressors

Historically burro deer have faced a range of threats from activities associated with an increasing human population in southeastern California. Development and agriculture along the Colorado River has reduced access to the summer riparian habitats, introduced invasive species such as salt cedar, and reduced the availability of native habitats. In addition, increased recreation development and flood control measures have contributed to reduced available summer habitat.

In areas away from the riparian lowlands, increased recreational use of desert washes by off-highway vehicles (OHVs) has resulted in localized disturbances of burro deer, and effectively has reduced connectivity between riparian and mountain habitats. Other localized impacts include mining operations and energy development (Celentano and Garcia 1984).

Historically, poaching, road kill, and drowning in canals have all been identified as significant sources of mortality, although measures taken to reduce road kill and drowning have had some success in reducing these mortality factors (CDFG 1995).

Competition from non-native grazing animals such as feral ass may represent a long-term pressure in shared habitat (Celentano and Garcia 1984; CDFG 1997). The most recent research confirms significant biological overlap in the diet of both species (Marshal et al. 2012).

Other threats found throughout the southwestern desert region include introduction of non-native pasture plants; overstocking and competition from cattle, domestic sheep, and goats; and extensive oil and gas development. However, as yet, these threats appear to be absent from the Southern California range of burro deer (Heffelfinger et al. 2006; Heffelfinger 2006).

Conservation and Management Activities

Several management activities have been implemented specifically to benefit burro deer, or for other species that also benefit the subspecies.

The 1984 *Burro Deer Herd Management Plan* (Celentano and Garcia 1984) was prepared in response to possible stressors and threats from development, agriculture, poaching, and OHVs. The management plan identified actions to maintain habitat health and connectivity as well as actions to mitigate known anthropogenic sources of mortality. The plan included the following key action points:

a) Maintain access to riparian habitats in summer by controlling recreational uses of riparian habitats, and ensuring agricultural practices are sympathetic to deer requirements.

b) Maintain contiguous access between summer riparian habitat and winter mountain habitats by ensuring desert wash systems are maintained and not fragmented by development.

c) Manage access of OHVs to desert wash habitats in core deer population areas.

d) Reduce road kill incidences along State Highways 78 and 95 by promoting the construction of fencing and underpasses that allow deer to travel between the Colorado River and mountainous habitats.

e) Ensure that artificial canal construction uses methods that reduce likelihood of deer drowning; e.g., implementation of 2:1 slopes, use of linear curbing.

f) Reduce illegal hunting.

g) Document the effectiveness of water source development, i.e., developing catchments that improve availability of free water. This serves two goals: (1) reduces the reliance of deer on open canals as a water source in the driest parts of the year, and thus reduces the risk of drowning; and (2) improves overall access to water for the wider herd.

Desert Wildlife Unlimited Inc. is also involved in providing and maintaining drinkers for desert wildlife, including burro deer. The organization employs 12,000-gallon fiberglass tanks with a step drinker attached, which require relatively little maintenance (Desert Wildlife Unlimited Inc. 2013).

While historically access to permanent water sources has been viewed as the most significant factor limiting desert wildlife, and improvement of water sources has therefore been a primary goal of conservation management (Celentano and Garcia 1984), water sources may only be a limiting factor in the hottest and driest seasons. Throughout much of the year, herd size limitations may be a function of available forage (Marshal et al. 2006b). More recent management recommendations have focused on methods for improving forage availability.

The burro deer should also benefit from habitat conservation and management measures being implemented by the Lower Colorado River Multi-Species Conservation Program (LCR MSCP 2004).

Although the burro deer is not a covered species under the LCR MSCP, one of the conservation measures in the LCR MSCP is to provide replacement riparian habitat, which would benefit burro deer, including removal of tamarisk and replacement with suitable native habitat. An LCR MSCP conservation goal is to create 765 acres of cottonwood-willow and honey mesquite vegetation.

Data Characterization

Burro deer are generally well studied, at least from the perspective of game management. The burro deer herd is managed for harvesting as part of the broader mule deer population in California. Because of its unique desert habitat and management needs, it is managed within its own Deer Management Unit (D12). Annual harvest records are collected from hunters and used in conjunction with fall herd composition data and spring surveys to predict the available bucks for the next hunting season (CDFG 2007, 2010).

Efforts to quantify burro deer population parameters, including population trends and health, have been more difficult because of low densities and low detection probabilities (Thompson and Bleich 1993). Celentano and Garcia (1984) provided estimates of herd density and habitat utilization, but identified a lack of long-term data pertaining to (a) herd age class and sex composition, (b) effects of predators, and (c) effects of illegal kills.

Subsequent studies largely focused on understanding herd composition and age structure (e.g., Thompson and Bleich 1993; Marshal et al. 2005, 2006c), and on quantifying the relationship between rainfall, forage quality, population fluctuations, and management activities (Marshal et al. 2002, 2006a, 2006b, 2012; Marshal and Bleich 2011). However, explicit studies examining the impacts of predators and poaching on this subspecies are absent from the scientific literature. Further, most of the recent studies have been focused in the east Chocolate–Cargo Muchacho areas, providing little information on the status of the herd across the entirety of its range.

Management and Monitoring Considerations

Ongoing management of burro deer herds includes actions to monitor and maintain habitat quality and connectivity as well as activities to reduce known sources of anthropogenic mortality:

- Management of development within riparian and xeroriparian habitats to ensure access between summer and winter ranges to riparian habitats and clear migration corridors along desert washes (Celentano and Garcia 1984; CDFG 1994, 1995).
- Ongoing monitoring of the effects of illegal hunting (CDFG 1995).
- Assessment and management of feral ass populations to reduce potential competitive effects (CDFG 1997).
- Assessment and development of alternative forage management and enhancement methods to improve quantity and quality of available forage (Marshal et al. 2006a).

Predicted Species Distribution in Plan Area

This section provides the results of habitat modeling for burro deer, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

The model generated 1,150,569 acres of modeled suitable habitat for burro deer within the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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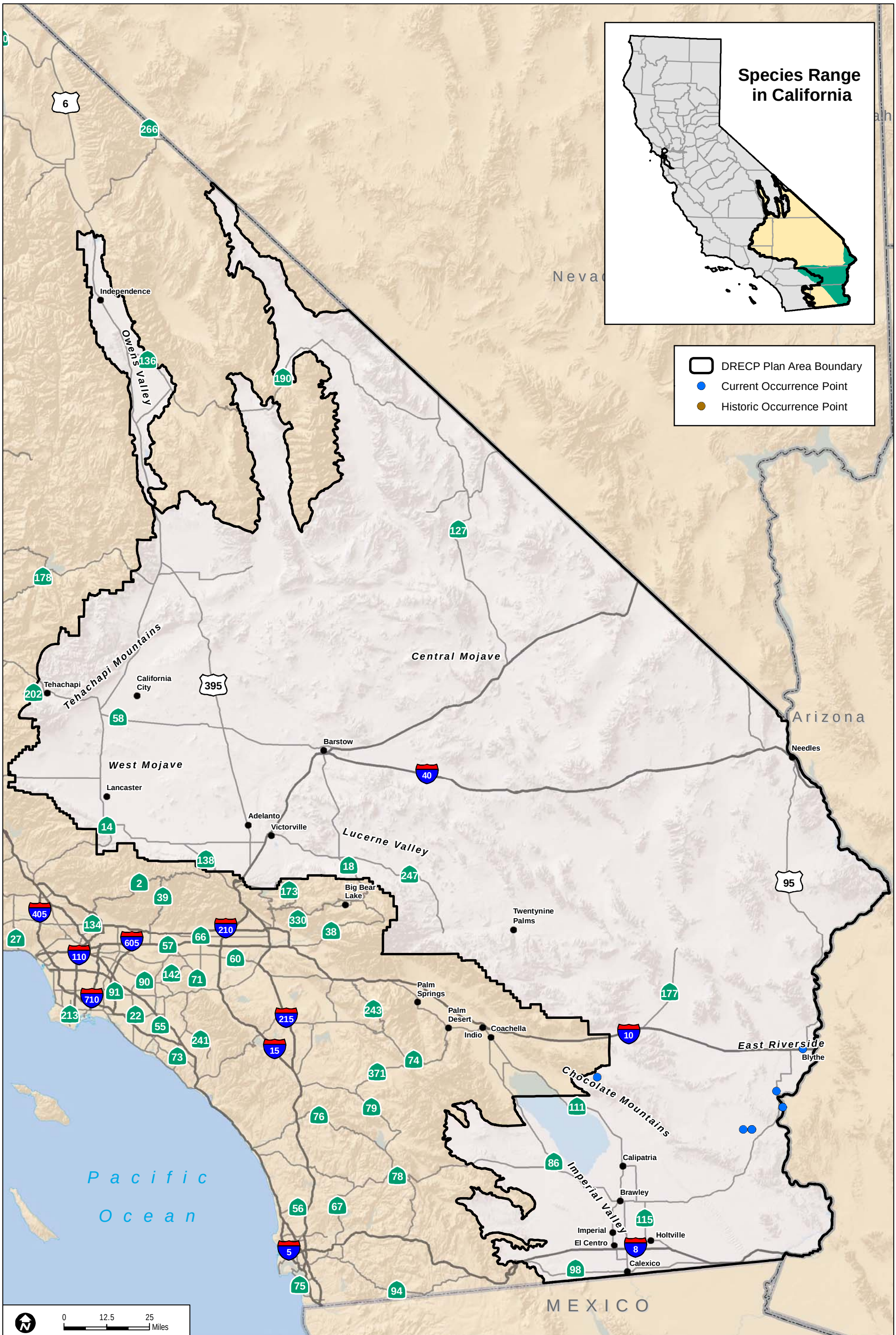

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M02

Burro Deer Occurrences in the Plan Area

California Leaf-Nosed Bat (*Macrotus californicus*)

Photo courtesy of Jason Corbett, Bat Conservation International, www.batcon.org.

Legal Status

State: Species of Special Concern
Federal: Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

The California leaf-nosed bat (*Macrotus californicus*) is in the family Phyllostomidae and was originally assigned as a distinct full species (Baird 1858, as cited by Rehn 1904). However, based on morphometrics, Anderson and Nelson (1965) placed California leaf-nosed bat as a subspecies of Waterhouse's leaf-nosed bat (*Macrotus waterhousii californicus*), and this was followed by others (e.g., Hall 1981). Based on cranial measurements and chromosomal and biochemical information, California leaf-nosed bat was reassigned to a separate full species *M. californicus* (Davis and Baker 1974; Davis 1973; Greenbaum 1975). Davis and Baker (1974) concluded that *M. californicus* and *M. waterhousii* are "parapatric" species that have contiguous, but non-overlapping distributions. *M. californicus* is currently accepted as a separate species (Wilson and Reeder 2005). A physical description of the species can be found in Wilson and Ruff (Brown 1999).

Distribution

General

The California leaf-nosed bat occurs from southern Nevada and Southern California east to Southern Arizona and south to northern Sinaloa, southwestern Chihuahua, Baja California, and Tamaulipas, Mexico (Wilson and Reeder 2005) (Figure SP-M03). In California, the California leaf-nosed bat occurs in the desert regions of eastern San

Bernardino (i.e., excluding the western Mojave region), Riverside, and San Diego counties and all of Imperial County (Brown and Berry 2004). Although historically the range of California leaf-nosed bats in California reached almost to the southern California coast (Los Angeles/Ventura County line; southern coastal San Diego County, Santa Margarita Ranch [now Camp Pendleton] and DeLuz), the species no longer occurs in these areas, despite repeated searches by bat biologists (Brown and Berry 1998, 2004). Roost disturbance and more important, the loss of suitable foraging habitat have probably led to this regional extirpation (see discussion under Threats and Environmental Stressors). However, even more recent texts do not recognize this loss of range in California in areas outside of the California desert regions that has occurred over the past 60 years (Harvey et al. 2011).

Distribution and Occurrences within the Plan Area

Historical

There are two historical (i.e., pre-1990) occurrences for the California leaf-nosed bat in the Desert Renewable Energy Conservation Plan (DRECP) Area located west of Yuma, Arizona, and north of Interstate 8 (I-8) (Grinnell 1918; Brown et al. 1993a; Brown and Berry 1998, 2004 and 2005; CDFW 2013; Dudek 2013). In writing the bat section of the Bureau of Land Management (BLM) California Desert Plan in 1980, Brown reviewed all historical literature and museum records for bats in the California desert and included her own observations since 1968. (These records occur in the CNDDB as supplied by BLM regardless of the original source.) Brown and Berry (1998, 2004) surveyed 18 historical sites (records more than 60 years old), and of these, 8 (45%) still sheltered California leaf-nosed bats at the time of the surveys. Howell (1920) also noted that this species was common in caves and mines and that the Salton Sea area supported many caves created by wave action of the sea along its historical coastline. Howell (1920) observed up to 300 individuals in a single colony and collected 63 of them. Arnold (1943) observed the species in the winter in mines and powder magazines near the Laguna and Imperial dams in Imperial County, and Huey (1925) observed a colony of about 500 individuals in a mine shaft north of Potholes in Imperial County. Several historical sites for California leaf-nosed bat occur in San Diego

County, including in the Plan Area at the Mollie Mine in Anza Borrego State Park and a natural cave in Flat Cat Canyon (Banks 1965), as well as the Stage Station at Vallecito and the Artery Mine near Dulzura (Krutzsch 1948) west of the Plan Area. Brown and Berry (1998) visited these areas during the 1980s and 1990s, when assessing the current range for California leaf-nosed bats for the California Department of Fish and Wildlife (CDFW), and no California leaf-nosed bats were found.

Recent

There are numerous recent (i.e., since 1990) records for the Plan Area, including 39 occurrences in the California Natural Diversity Database (CNDDB) (CDFW 2013) and four roost sites (Figure SP-M03). Brown (pers. comm. 2012) also has provided many records for California leaf-nosed bat in the California desert region. Brown has surveyed more than 2,500 mines or natural caves in 30 mountain ranges in the desert within the range of California leaf-nosed bat over the past 45 years (Brown 1993; Brown and Berry 1998, 2000, 2004). Mountain range extensions (beyond museum and past literature citations) for this species included the Bristol, Marble, Calumet, Eagle, Pinto, Ship, Old Woman, McCoy, Sacramento and Little Maria Mountains in Riverside and San Bernardino counties. Warm mines (and California leaf-nosed bat) have yet to be discovered in other adjacent mountain ranges (Orocopia, Chuckawalla, Little Chuckawalla, Palen, Granite, Coxcomb, Arica, West Riverside, Turtle, Sawtooth, Piute, Clipper, Sheephole and Stepladder Mountains). During a 1995 survey conducted for the Fort Irwin Expansion (Brown and Berry, unpublished data, as cited by Brown, pers. comm. 2012), a few male California leaf-nosed bats were discovered in May in the "Mud Hills" mine at the north edge of the Avawatz Mountains, just south of Death Valley National Park. Guano attributable to this species was also located in a mine near Amargosa Springs. These records suggest a northward extension of the range of California leaf-nosed bat, and the species might occur in the southern part of Death Valley National Park (Brown, pers. comm. 2012).

Natural History

Habitat Requirements

In the California desert, all of the known California leaf-nosed bat roosts are located below 800 meters (2,500 feet) in elevation and most are within 6 kilometers (4 miles) of desert washes containing ironwood (*Olneya tesota*), palo verde (*Parkinsonia* spp.), smoke trees (*Psorothamnus spinosus*) and/or desert willows (*Chilopsis linearis*) (Brown, pers. comm. 2012). The greatest concentration of roosts and those with the largest bat colonies are within the drainage of (and often within sight of) the Lower Colorado River. The roosts discovered near the south end of Death Valley are located in creosote bush scrub. Historical roosts (before development) near coastal areas of California were in chaparral or oak woodland (Brown, pers. comm. 2012).

The California leaf-nosed bat is primarily a cave and mine dwelling species (Anderson 1969; Arita 1993; Arnold 1943; Brown and Berry 2003, 2004; Howell 1920), but also occasionally occupies buildings (Anderson 1969). In Arizona, they have also been found in "open" bridge structures that have cave-like chambers at either end (Davis and Cockrum 1963; Brown and Berry 2004), but most bridge structures are unlikely to be suitable as day roosts. California leaf-nosed bats have been observed using buildings as night roots east of Searchlight, Nevada (Hatfield 1937) and at Cibola National Wildlife Refuge in California (Brown and Berry 2003). Most winter roost sites in California are mine tunnels at least 100 meters (328 feet) long (Brown 2005). Roost chambers often have large ceilings and considerable fly space (Anderson 1969), although smaller drifts are also used. California leaf-nosed bat is the most northerly representative of the Phyllostomidae, a predominantly Neotropical family. This species neither hibernates nor migrates, and it is incapable of lowering its body temperature to become torpid. Bell et al. (1986) conducted a series of experiments in the laboratory to measure energy metabolism, thermoregulation and water flux to determine if special physiological adaptations allowed California leaf-nosed bats to remain active yearlong in the temperate zone. In the field, daily energy budgets for free-ranging bats were determined using the doubly-labeled water technique. California leaf-nosed bat has a relatively narrow thermal

neutral zone, with the lower critical temperature near 34 degrees Celsius (93 degrees Fahrenheit) and the upper near 37 degrees Celsius (98.6 degrees Fahrenheit). No special physiological adaptations were found in California leaf-nosed bat for desert existence (Lu and Bleier 1981), and they appear to adapt behaviorally rather than physiologically by roosting in geothermally heated winter roosts that have a stable year-round temperature of about 27 degrees Celsius (81 degrees Fahrenheit) (Bell et al. 1986; Brown 2005; Brown and Berry 1998, 2004). Summer roosts may be in more shallow natural rock caves and mines since the summer desert temperatures close to the openings exceed 40 degrees Celsius (104 degrees Fahrenheit) (Brown 2005). Summer roost sites are not always completely dark, and individuals may roost within 10 to 30 meters (33 to 98 feet) of the roost opening. California leaf-nosed bats are tolerant of the highly ammoniated atmosphere of many caves and mines and can tolerate higher concentrations than humans (Mitchell 1963).

California leaf-nosed bats forage in riparian and desert wash areas in California, Arizona, and Nevada (Brown 2005; Huey 1925; Williams et al. 2006) and at tinajas (water-carved natural rock pools) and manmade tanks in southwestern Arizona (Rabe and Rosenstock 2005; Schmidt 1999). Williams et al. (2006) observed California leaf-nosed bats generally using riparian marsh, mesquite bosque, riparian woodland, and riparian shrubland without any apparent differential selection. The tinajas in the Rabe and Rosenstock (2005) study provided open flight approaches and were located near suitable roosting sites (cliffs and rocky canyons). For California, suitable foraging habitats are desert riparian, desert wash, desert scrub, desert succulent scrub, alkali desert scrub, and palm oases (Brown and Berry 2004; Zeiner et al. 1990). In the Sonoran Desert of Arizona (where desert trees are not confined to drainages), a greater percentage of the landscape is utilized by foraging bats (Brown et al. 1999; Dalton et al. 2000; Dalton 2001).

Roosting and foraging habitat associations for the California leaf-nosed bat in the Plan Area are shown in Table 1.

Table 1. Habitat Associations for California Leaf-Nosed Bat

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Mines and Caves and occasionally buildings	Roosting	Mines within the California Wildlife Habitation Relationship distribution map boundaries.	Anderson 1969; Zeiner et al. 1990; Brown and Berry 2004
Riparian woodlands desert wash, desert scrub	Foraging	Riparian woodlands, desert wash, desert scrub within 6.2 miles of mines.	Williams et al. 2006; Zeiner et al. 1990; Brown and Berry 2004

Foraging Requirements

California leaf-nosed bat appears to be primarily insectivorous (Anderson 1969). Prey for California leaf-nosed bat include Orthoptera (crickets and grasshoppers), Lepidoptera (butterflies and moths), Coleoptera (beetles), Homoptera (cicadas), and Hymenoptera (ants) (Anderson 1969; Huey 1925; Ross 1961), but at least occasionally takes small vertebrates. Brown (Brown and Berry 2003, 2004) discovered a California leaf-nosed bat in a night roost chewing on the head of a wiggling tree lizard (*Urosaurus ornatus*). Since that time Brown has seen other California leaf-nosed bats carrying tree lizards into night roosts. This reptile spends most of its time in trees and scrubs, often clinging head downward (Stebbins 1985). The California leaf-nosed bat probably gleaned it from the branches of a desert tree when the lizard was sleeping. They are vegetation gleaners and likely take prey directly from the ground or vegetation because some of their prey are flightless and sometimes diurnal (butterflies and lizards) (Stager 1943; Brown and Berry 2004; Anderson 1969; Bell and Fenton 1986). They have short, broad wings that allow them to fly slowly while foraging, with high maneuverability (Anderson 1969; Vaughan 1959), but they are also capable of fast flight with measured speeds of 12 to 14 miles per hour (Dalton 2001; Hayward and Davis 1964). They probably use a combination of echolocation, prey-produced sounds, and binocular vision to locate terrestrial prey (Bell 1985; Bell and Fenton 1986). Their eyes are positioned more anteriorly, and they have superior vision compared to other bats (Bell and Fenton 1986). They usually emerge from day roosts 90 minutes to

2 hours after sunset during the summer and forage in two main bouts during the night (Anderson 1969). During the winter, they may emerge around sunset or shortly after (e.g., within 30 minutes) and forage for about 2 hours (Brown 2005). They may use night roosts that are different from their day roosts (Anderson 1969; also see Hatfield 1937 for use of buildings as night roosts). In the summer, they will roost in desert trees with the foraging area as determined by radio-telemetry (Brown et al. 1999; Dalton et al. 2000).

Reproduction

The largest roosts (over 1,000 individuals of both sexes) are formed in the winter in warm mines. Segregation of males and females usually occurs in the spring and summer, although a few males remain in the maternity colonies. Females congregate in large (>100 bats) maternity colonies, although colonies of only 6 to 20 bats are also found (Barbour and Davis 1969; Vaughan 1959; Brown and Berry 2004). They utilize different mines or areas within a mine separate from those occupied in the winter. Within the larger colonies, clusters of five to 25 females will be associated with a single "harem" male that defends the cluster against intruding males (Brown and Berry 1991). The single young (weighing 25-30% of the mother's mass) is born between mid-May and early July (following a gestation of almost 9 months) and young are weaned by August (Anderson 1969; Bleier 1975; Bradshaw 1962; Carter and Bleier 1988; Brown and Berry 2004). Since the newborn bats are poikilothermic (a body temperature that fluctuates with the immediate environment), the maternity colony occupies areas close to the mine or cave entrance, where temperatures exceed 32 degrees Celsius (90 degrees Fahrenheit) and daytime summer outside temperatures reach over 49 degrees Celsius (120 degrees Fahrenheit). Most maternity roosts have multiple entrances that allow warm air flow through the mine.

Maternity colonies disband once the young are independent in late summer and breeding occurs in the early fall (Anderson 1969; Brown and Berry 1996). The reproductive cycle of these bats as studied by Krutzsch and others (Krutzsch et al. 1976; Crichton and Krutzsch 1985; Bodley 1974; Bleier 1975; Bradshaw, 1962) shows that viable sperm is not present in the male reproductive tract until August. Ovulation occurs in September and October (Bleier 1971), and unlike many other

bat species that store sperm over the winter and delay fertilization, fertilization occurs immediately after mating, and implantation occurs in later October and November to January (Bleier 1971; Carter and Bleier 1988). Gestation is 8 to 9 months and includes about a 4.5-month diapause period when growth and development is slowed (Bleier 1971; Bleier and Ehteshami 1981; Bradshaw 1962; Crichton and Krutzsch 1985; Crichton et al. 1990). Growth rate and diapause is under control of the hormone progesterone (Crichton and Krutzsch 1985; Crichton et al. 1990). In March, with increased temperatures and insect availability, embryonic development accelerates. Females are reproductively active in their natal year, but males become sexually mature in their second year (Carter and Bleier 1988). Longevity is at least 15 years, based on banding studies (Brown 2005).

In the fall, males aggregate in display roosts and attempt to attract females with a courtship display consisting of wing flapping and vocalizations . The areas used as "lek" sites are usually in or near a mine that had been occupied by a maternity colony (Berry and Brown 1995; Brown and Berry 2004), although exceptions exist. The lek site at Cibola Bridge is located over 11 kilometers (7 miles) from the roost at the Hart Mine (Brown and Berry 2003). In some mines, males defend specific calling areas, while at other sites they will display alongside other males. Aggression between males occurs at this time. Females enter the areas throughout the night, usually roosting in separate groups before approaching a male (Berry and Brown 1995). A banded male observed in the Queen Mine in the Cargo Muchacho Mountains (Imperial County) in September 1994 did not leave the mine during the night, and copulated with at least four females during this period (Brown, pers. comm. 2012). Since the majority of roost surveys have been conducted in the winter and summer, the fall courtship areas for California leaf-nosed bats have not been determined for most mountain ranges.

Key seasonal periods for the California leaf-nosed bat are summarized in Table 2.

Table 2. Key Seasonal Periods for California Leaf-Nosed Bat

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Reproduction				x	x	x	x	x				
Mating									x	x		
Wintering	x	x	x								x	x

Notes: Seasonal migration may occur between mountain ranges.
Sources: Anderson 1969; Bleier 1975; Bradshaw 1962; Brown and Berry 2004

Spatial Activity

California leaf-nosed bats are year-long residents in California (Anderson 1969; Brown and Berry 2004), although historically the species may have migrated to Mexico in the winter (Grinnell 1918) prior to the availability of abandoned mines. Bell et al. (1986) concluded that behavioral adaptations such as foraging methods and roost selection contributed to the successful exploitation of the temperate zone desert by California leaf-nosed bat.

The annual mean temperature in the California desert in the range of California leaf-nosed bat is approximately 23 degrees Celsius (73 degrees Fahrenheit) and the mean winter temperature is 14 degrees Celsius (57 degrees Fahrenheit). All known winter roosts in the deserts of California, Arizona and southern Nevada exhibit stable temperatures greater than 27 degrees Celsius (81 degrees Fahrenheit) and relative humidities above 22%. These mines appear to be located in geothermally-heated rock formations of moderate temperature (Higgins and Martin 1980). California leaf-nosed bats inhabit a stable warm environment (except during their short winter foraging periods). Roost site use does vary seasonally, however, with mixed male/female roosts in the winter and mostly segregated, large, female maternity roosts and smaller, dispersed male roosts during the spring through summer reproductive season (Anderson 1969; Brown 2005), indicating at least local seasonal movements and roost use related to reproduction. Banding studies conducted over the past 43 years suggest that distances traveled between summer and winter roosts are generally no more than a few miles (Brown et al. 1993b;

Brown and Berry 1996). Over 25,000 California leaf-nosed bats from mine roosts along the Colorado River from Parker Dam to Yuma were banded. On yearly trips, usually in the winter, many of these bats were recaptured up to 10 times with an average 50% recapture success rate, suggesting strong roost fidelity, although seasonal movements do occur between roosts. The longest distance between the site of banding and that of recapture was a movement over two mountain ranges for a linear distance of 87 kilometers (54 miles). The greatest time interval so far between initial banding and recapture is 15 years. Assuming that the bat was born in the spring prior to the winter banding, this would indicate a possible longevity of at least 15.5 years. This record for the species is remarkable because long life in bats is usually attributed in some part to their ability to undergo daily and seasonal torpor (Brown, pers. comm. 2012).

There is some information about spatial activity related to foraging. Vaughan (1959) reported that California leaf-nosed bats forage up to 1.3 kilometers (1 mile). Using radiotelemetry, Brown et al. (1993b) observed foraging in desert wash within 10 kilometers (6.2 miles) of roost sites. although more recent data documents captures of California leaf-nosed bats in cottonwood and willow revegetation sites along the Lower Colorado River over 16 kilometers (10 miles) from any potential roosting habitat (Calvert 2009a, 2009b, 2010). As observed by Williams et al. (2006), they generally forage in riparian habitats without any apparent differential selection of riparian type. They also forage at open water sites near potentially suitable roosting habitat (Rabe and Rosenstock 2005). Their ability to fly fast suggests that they could forage fairly far from roost sites. In addition, their selection of limited roosting areas (i.e., primarily temperate caves and mines) suggests that they may be capable of flying quite far to suitable foraging areas that support abundant insect prey, even if most activity is near roost sites (e.g., Williams et al. 2006).

Night roosts are occupied by California leaf-nosed bats between foraging bouts, and may have social significance to the colony. Night roosts are often identified by large amounts of guano and culled inedible insect remains (lepidopteran and orthopteran wings). Bats may return to the same mine used during the day, and roost in different areas. Radio-telemetry studies have shown that individual bats have fidelity to certain night roost sites in shallow mines, rock

shelters, buildings, bridges and trees (Brown et al. 1993b; 1999; Brown and Berry 2003; Dalton et al. 2000).

Ecological Relationships

There is some information about ecological associations for the California leaf-nosed bat, but little data for direct or indirect interspecific interactions. It can be found in association with other bat species at roost sites, including pallid bat (*Antrozous pallidus*), Townsend's big-eared bat (*Corynorhinus townsendii*), and myotis species (*Myotis* spp.) in California (Vaughan 1959; Brown and Berry 2003, 2004). Pallid bats and California leaf-nosed bats have similar ecological attributes as both glean large immobile insects and arthropods, and day and night roost in close proximity in mines. Pallid bats cluster in roosts and often use crevices, while California leaf-nosed bats hang alone from the ceiling (Vaughan 1959).

Desert riparian communities are very spatially limited resources used by a large number of bat species. A likely important factor in bat community diversity and ecological relationships in desert riparian areas is resource partitioning. Black (1974) suggested that bats may employ several types of foraging and food partitioning mechanisms that could reduce interspecific competition, including size and type of prey; periods of activity (most bat prey are active within a few hours of sunset, but different prey have different peak activity periods); spatial partitioning, such as between-, within-, and below-canopy foragers; and flight patterns, such as slow vs. fast flying, maneuverability, and hovering. Williams et al. (2006) examined foraging activity by California leaf-nosed bats in riparian habitats in southern Nevada that were also used by 14 other bat species, including both resident and migrant species (see Table 1 in Williams et al. 2006 for the list of species detected). Adequate detection data were collected to analyze habitat use by several of the species. These data show that California leaf-nosed bat, Brazilian free-tailed bat (*Tadarida brasiliensis*), western yellow bat (*Lasiurus xanthinus*), and pallid bat exhibit different habitat selection patterns. While California leaf-nosed bat and Brazilian free-tailed bat were riparian habitat generalists, western yellow bat and pallid bat showed strong preferences for riparian woodland (Williams et al. 2006). Six other bats qualitatively showed more activity in one of the four riparian

types (i.e., riparian marsh, mesquite bosque, riparian woodland, and riparian shrubland), indicating some selection. Overall, riparian woodland, which represented less than 1% of the riparian habitat in the study area, was the preferred habitat type (>50% of all bat activity), with riparian marsh the least used, although it was often used by the spotted bat (*Euderma maculatum*). Williams et al. (2006) suggested that habitat preferences by the different bats may reflect preferred insect prey and abundance, indicating a possible basis for resource partitioning. Given that desert riparian communities are a critical resource for bats, the habitat use information provided by Williams et al. (2006) indicates that managing this diverse habitat type, including hydrology and species composition, is important for maintaining a diverse bat community, including suitable habitat for California leaf-nosed bat.

Population Status and Trends

Global: Apparently secure (NatureServe 2011)
State: Vulnerable to imperiled (CDFG 2011)
Within Plan Area: Same as state

Although historical records from 1894 through 1950 place California leaf-nosed bat in more coastal sections of southern California, these sites are not currently occupied (Grinnell 1918; Howell 1920; Constantine, 1961, 1998; Brown and Berry 1998, 2004), representing a loss of almost 50% when polygons are drawn between historical and current roost areas in California. Urbanization, human disturbance of roosts and destruction of foraging areas are probably the primary factors in their eradication from these areas. With possibly one exception, all California leaf-nosed bat roosts are now located in the desert.

The California leaf-nosed bat is a former U.S. Fish and Wildlife Service (USFWS) Category 2 Candidate for listing under the federal Endangered Species Act and is now a Species of Special Concern for USFWS and the CDFW (Brylski et al. 1998), and a BLM and U.S. Forest Service (Region 5) Sensitive Species. The Western Bat Working Group granted it High Priority for its entire range. www.wbwg.org/ speciesinfo/species_matrix/spp_matrix.pdf).

Information collected by Ellison et al. (2003) for California leaf-nosed bat suggested that assessing population trends for this species would be a challenge. Ellison et al. (2003) reviewed information for 143 locations in Arizona, Nevada, and California. Counts at occupied sites ranged from 1 to 2,000 individuals. Trends were analyzed for five colonies, including three winter colonies and two summer colonies, and no positive or negative population trend was apparent. They also noted that the number of individuals at roost sites can fluctuate both between and within seasons, so population sampling would need to account for this apparent natural temporal variation. Ellison et al. (2003) noted, however, that many reports lacked careful and consistent documentation of surveys methods, such as how counts were made, what type the colony was, etc. More recent censuses using standardized methods has revealed stable colony sizes for California leaf-nosed bats in the largest colonies. Over the last 10 to 12 years Brown has conducted censuses by counting exiting bats in the evenings with night vision equipment in the same manner and at the same times of year in the absence of moonlight (Brown 2011). These are usually done in the winter (January or February) when the largest colonies form and for maternity colonies in mid-April or May (prior to young of the year flying). Moon phase was recognized as a significant variable in determining population size by exit counts for California leaf-nosed bat in January 2003 when paired counts were conducted during the week before and after the full moon on selected mines in southeastern California (Brown and Berry 2004; Brown 2011). There was a several-fold increase in the number of bats exiting the mine in the hour after dark in the absence of moonlight. These studies by Brown underscore the need for standardized census methods and consideration of detectability factors to document any population trends.

Threats and Environmental Stressors

The two main threats to this species likely are (1) disturbances of roost sites due to human entrance, abandoned mine closures, and renewed mining in historic districts (Brown 2005; Zeiner et al. 1990) and (2) loss and degradation of desert riparian habitats (Brown 2005). Brown (Brown 2005; Brown and Berry 1998, 2004) cites the loss of desert riparian habitat to development of golf courses and residential housing

in the Coachella Valley and the "rip rapping" and channelization of desert washes as a threat to the species. Ground water pumping and road construction that alters drainage patterns can negatively impact microphyll woodland and desert wash vegetation. Another potential threat is direct or secondary poisoning and loss of prey related to pesticide use for agriculture and golf course operations, and other environmental contaminants associated with mining (Clark 1981; Clark and Hothem 1991).

Several recent studies have documented substantial mortality of bats at wind facilities (e.g., Baerwald and Barclay 2009; Cryan 2011; Cryan and Barclay 2009). A general review of the wind facility–related literature failed to reveal evidence for, or discussions of, California leaf-nosed bat fatalities or assessed risks at wind facilities (e.g., Baerwald and Barclay 2009; Cryan 2011; Cryan and Barclay 2009; Cryan and Brown 2007; Kuvlesky et al. 2007). This is likely because of the species' limited range in the southwestern United States and, further, because relatively little systematic post-project bat fatality monitoring data have been collected for large wind energy projects in the southwest (Solick and Erickson 2009). However, California leaf-nosed bats in the Plan Area could be at elevated risk of turbine strikes or from other associated causes (e.g., barotrauma) if a wind facility was located within a few miles of a day roost site (where most foraging activity occurs) and strikes would most likely occur during emergence and return to the day roost. Risk of strikes may also be higher when bats are moving between maternity roosts and winter sites in the fall and spring.

Conservation and Management Activities

California leaf-nosed bat is addressed in the West Mojave Plan (BLM 2005) under Alternative A (the Proposed Action – Habitat Conservation Plan). The BLM would implement several conservation measures for California leaf-nosed bat, including:

- Protection of all roosts containing more than 10 California leaf-nosed bats (Notes: The Plan identified one maternity roost and one maternity/winter roost for the species. Also, the Plan refers to "maternity and hibernation" roosts, but California leaf-nosed

bats do not hibernate (Brown, pers. comm. 2012) so reference to these roost types was deleted);

- Continued fencing around (but not over) open, abandoned mine features to provide bats access to roosts and to reduce hazards to the public;
- Required surveys for bats by applicants seeking discretionary permits for projects that would disturb natural caves, cliff faces, mine features, abandoned buildings, or bridges to determine whether significant roost sites are present; and
- Safe eviction of bats at a non-significant roost (i.e., fewer than 10 individuals) prior to disturbance or removal.

BLM would also conduct monitoring and adaptive management for California leaf-nosed bats. Monitoring actions include:

- Determining bat numbers in all significant roosts (defined by BLM for the West Mojave Plan as more than 10 individuals);
- Conducting periodic surveys of mine openings in Pinto Mountains for bats in areas with high potential for containing significant roost sites;
- Determining and reporting the effectiveness of mitigation measures providing for safe exit of bats;
- Reporting take from approved projects that impact bats under to the CDFG and USFWS; and
- Monitoring population numbers using bat houses if installed (Note: Brown (pers. comm. 2012) indicates that California leaf-nosed bats would not use bat houses, but this is included as conservation measure in the West Mojave Plan).

Adaptive management measures include:

- Gating mines where new significant roosts are found;
- Installing bat houses in locations, where appropriate, if populations decline or are threatened (Note: Brown (pers. comm. 2012) indicates that California leaf-nosed bats would not use bat houses); and

- Desert wash vegetation within 3 miles of known or newly discovered maternity and hibernation roosts of California leaf-nosed bats would be protected. Motorized vehicle use of washes in these locations would be assessed on a case-by-case basis to determine if vehicles harm the desert wash vegetation. If substantial damage from vehicle use is determined to be present, alternative access routes would be developed and the wash routes would be closed or limited. (Note: California leaf-nosed bat does not hibernate (Brown, pers. comm. 2012), but the West Mojave Plan refers to hibernation roosts).

The California leaf-nosed bat is also addressed in two other BLM plans for the California desert. The *Proposed Northern and Eastern Mojave Desert Management Plan* addresses sensitive bats, including California leaf-nosed bat (BLM 2002a). Under the proposed alternative, this plan includes changing the existing "Moderate Multiple Use Classification" to the "Limited" designation for 7,400 acres of public land in the Silurian Hills region, which is known to support extensive habitat for several sensitive bat species. Route designation would occur on these lands, including seasonal limitations and/or closures to sensitive bat values (e.g. active bat maternity roosts).

The *Proposed Northern & Eastern Colorado Desert Coordinated Management Plan Activities* (BLM 2002b), under all alternatives, would require mitigation measures for projects authorized at or within 1 mile of a significant bat roost site, which may include seasonal restrictions, light abatement, bat exclusion, and gating of alternate sites. If bats are to be excluded from an old mine prior to renewed mining, the exclusion must be performed at a non-critical time by a qualified bat biologist. Mitigation plans for large mines would consider retaining some shafts and adits (horizontal or nearly horizontal opening to a mine) or creating new ones as compensation. Also, under the proposed alternative, Bat gates would be constructed on caves or mine roosts only where there is significant potential for negative effects and closure of any route within 0.25 mile of any significant bat roost would be strongly considered.

In addition, as a BLM sensitive species, California leaf-nosed bat is addressed under other land use actions undertaken by BLM. In

accordance with BLM's "6840 – Special Status Species Management" manual, the objectives for sensitive species policy are:

> To initiate proactive conservation measures that reduce or eliminate threats to Bureau sensitive species to minimize the likelihood of and need for listing of these species under the ESA (BLM 2008).

Under this policy, BLM must consider the impact of actions on sensitive species, including outcomes of actions (e.g., land use plans, permits), strategies, restoration opportunities, use restrictions, and management actions necessary to conserve BLM sensitive species.

The California leaf-nosed bat is covered as an "evaluation species" under the Lower Colorado River Multi-Species Conservation Program administered by the Bureau of Reclamation (LCR MSCP 2004). The LCR MSCP defines evaluation species as species that could be listed in future years and that could be added to the covered species list during LCR MSCP implementation, but for which sufficient information was not available for LCR MSCP planning area when the plan was prepared. Conservation measures include: (1) conducting surveys for roost sites within 5 miles of the LCR MSCP planning area in Reaches 3–5; and (2) creating habitat near roost sites, including cottonwood-willow and honey mesquite within 5 miles of roost sites.

California leaf-nosed bat is also addressed in the Military Integrated Resource Management Plan (INRMP) for the Marine Air Ground Task Force Training Command, Marine Corps Air Ground Combat Center, Twentynine Palms (MAGTFTC MCAGCC 2007). As a designated sensitive species in the INRMP, California leaf-nosed bat is provided protection and management considerations for the military training operations at Twentynine Palms. If it is determined to be at risk from training activities, efforts are made to avoid and minimize impacts. For example, four bat gates have been installed in three mines to allow bats access to roosts without disturbance from humans. The Twentynine Palms INRMP also includes three objectives:

- Monitoring current bat gates to inspect for trespass and condition;
- Evaluating mine entrances for installation of bat gates to those mines that are exceptional bat habitat but not culturally significant; and

- Evaluating modification of bighorn sheep guzzlers for use by bats and other wildlife to enhance habitat value.

Data Characterization

There is substantial information for the distribution of California leaf-nosed bat and its use of mines and caves in the Plan Area. Brown has surveyed more than 2,500 mines or natural caves in 30 mountain ranges in the desert within the range of California leaf-nosed bat over the past 45 years (Brown 1993; Brown and Berry 1998, 2000, 2004).

Management and Monitoring Considerations

The main management consideration for California leaf-nosed bat is the relationship between human activities near active roost sites, (mine entry by recreation, geologists, etc.), and mine closure for hazard abatement or renewed mining (Brown 2005). Removal of desert wash vegetation near a roost will cause declines (Brown and Berry 1995). Management of riparian communities with regard to hydrology and community structure is also an important management concern (Williams et al. 2006). Pesticide use in agricultural areas or golf courses adjacent to suitable roosting and foraging areas should be managed to prevent potential direct and indirect poisoning and secondary impacts on prey.

Predicted Species Distribution in the Plan Area

This section provides the results of habitat modeling for California leaf-nosed bat, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 8,046,536 acres of modeled suitable habitat for California leaf-nosed bat in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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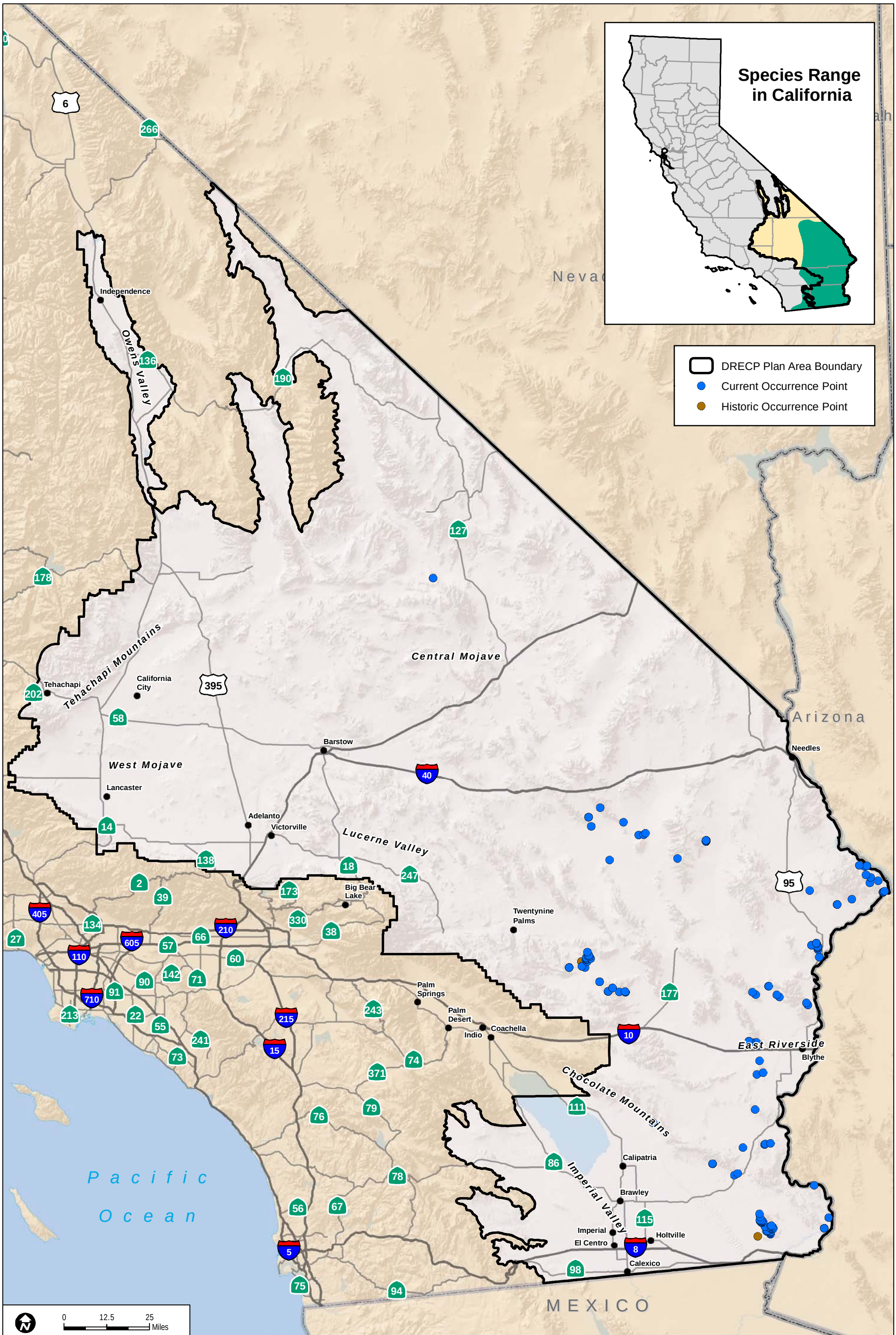

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M03

Leaf-nosed Bat Occurrences in the Plan Area

Desert Bighorn Sheep (*Ovis canadensis nelsoni*)

Photo by Dee E. Warenycia.

Legal Status

State: None for subspecies *Ovis canadensis nelsoni* (Nelson's bighorn sheep); Peninsular bighorn sheep distinct population segment (DPS) is Threatened, Fully Protected

Federal: Peninsular bighorn sheep DPS is Endangered; Nelson's bighorn sheep is Bureau of Land Management Sensitive, U.S. Forest Service Sensitive

Critical Habitat: Designated for Peninsular bighorn sheep DPS occupying the Peninsular Ranges of Southern California on April 14, 2009 (74 FR 17288–17365).

Recovery Planning: A Recovery Plan for Peninsular bighorn sheep in the Peninsular Ranges of California was approved October 25, 2000 (USFWS 2000).

Taxonomy

The subspecific taxonomy of bighorn sheep (*Ovis canadensis*) at the subspecies level in the southwest desert region has been uncertain. Earlier studies had placed desert bighorn sheep in one of four subspecies occurring in the southwest desert region (Cowan 1940). For populations within the Desert Renewable Energy Conservation Plan (DRECP) Area, based on cranial measurements, desert bighorn sheep in the Peninsular Ranges were considered a separate subspecies, *O. c. cremnobates,* and northerly populations were designated *O. c. nelsoni* (Nelson's bighorn sheep). More recent genetic and morphometric information does not support the distinct subspecific delineation of *O. c. cremnobates* and the current classification has Nelson's bighorn sheep as the only bighorn subspecies occurring in the Plan Area. Research has found north-south and elevational variation in life history patterns of Nelson's bighorn sheep that tracks differences in temperature regimes in California and on a larger geographic scale (Wehausen 2005, 2006)

but with no clear boundaries that might be used to define subspecies. This clinal variation supports Ramey's (1995) suggestion that all desert bighorn sheep be recognized as one polytypic subspecies. Wehausen (2006) suggested that such regional variation be recognized and considered in conservation planning.

In the 2009 federal critical habitat designation, desert bighorn sheep in the Peninsular Ranges are treated as a DPS of the Nelson's bighorn sheep, and are no longer referred to as a separate subspecies (74 FR 17288–17365). This DPS is federally listed as endangered and state-listed threatened and fully protected. Consistent with the federal critical habitat designation, the common name Peninsular bighorn sheep is retained in this species profile where the information pertains specifically to the federally and state-listed DPS. The common name desert bighorn sheep is used elsewhere where this distinction is not made, but this information for desert bighorn sheep would also apply to the Peninsular bighorn sheep DPS.

Distribution

General

Desert bighorn sheep occur in the desert mountain ranges from the White Mountains in Mono and Inyo counties, south to the San Bernardino Mountains, then southeast to Mexico (Wehausen 2006; Shackleton 1985) (Figure SP-M01). An isolated population occurs in the San Gabriel Mountains (Zeiner et al. 1990). Beyond California, its range extends into southern Nevada, southern Utah, southwestern Arizona, and northwestern Mexico and Baja California, Mexico (Shackleton 1985). Although desert bighorn sheep has a broad overall geographic range, actual populations within the range are scattered and discrete (Shackleton 1985).

The Peninsular bighorn sheep DPS generally occurs in the Peninsular Ranges from the San Jacinto and Santa Rosa ranges south into Mexico. The DPS critical habitat is located in Riverside, San Diego, and Imperial counties (74 FR 17288–17365). The bighorn sheep in this region are restricted to the east-facing, lower elevation slopes below about 1,400 meters (4,593 feet), and most occur at elevations between 91 and 1,219 meters (300 and 4,000 feet) (63 FR 13135).

Distribution and Occurrences within the Plan Area

Historical

All of the California Natural Diversity Database (CNDDB) occurrences of desert bighorn sheep, excluding the Peninsular bighorn sheep DPS, within 5 miles of the Plan Area are historical (i.e., before 1990). These occurrences range from the Last Chance Range near the northeastern portion of the Plan Area south to the Chocolate Mountains in the southeastern portion of the Plan Area. Records marking the eastern boundary of the CNDDB records are from near Straw Peak, the Newberry Mountains, and the San Bernardino Mountains east of Joshua Tree National Monument (CDFW 2013).

Five of the six CNDDB records for Peninsular bighorn sheep within 5 miles of the Plan Area are historical. All of these records lie west of the southern portion of the Plan Area, three are within Anza-Borrego Desert State Park, one is near In-Ko-Pah Gorge, and one is east of San Bernardino National Forest (CDFW 2013).

Recent

The California Department of Fish and Game (CDFG)(2010a) prepared the *Biennial Report to the Legislature Regarding Desert Bighorn Sheep Management* pursuant to Section 4094 of the California Fish and Game Code. This report summarizes census information related to long-term management of desert bighorn sheep (including the authorization of hunting tags) and includes sheep counts in specific management units in 2009 and 2010. The distribution of desert bighorn sheep is grouped by a regional system of subpopulations (or metapopulations) based on natural physical features such as geography and vegetation that affect species occurrence, as well as manmade obstacles that affect distribution, such as freeways (CDFG 2010a). Aerial surveys in 2009 and 2010 documented 1,022 desert bighorn sheep, including ewes, lambs, and rams, in the following mountain ranges: Marble Mountains; Clipper Mountains; Kelso Peak and Old Dad Peak; Clark, Kingston, and Mesquite Mountains; Orocopia Mountains; Sheephole Mountains; South Bristol Mountains; Cady Mountains; White Mountains; and San Gorgonio Mountains. The 1,022 individuals represent minimum populations in these areas because

they were the only animals actually observed; population size is assumed to be larger (CDFG 2010a). The CDFG (2010a) report included the Peninsular bighorn sheep metapopulation, with an estimate of about 950 adults and recruited lambs among the nine distinct subpopulations as of December 2010. Population sizes and trends throughout the species' range in the Plan Area are discussed in more detail in the "Population Status and Trends" subsection.

There are 35 recent occurrences of the Peninsular bighorn sheep DPS in the Plan Area and 13 occurrences just west of the Plan Area (Dudek 2013). These occurrences are clustered in the extreme southwestern portion of the Plan Area (Figure SP-M01).

Natural History

Habitat Requirements

Desert bighorn sheep are mobile and wide-ranging and require a variety of habitat characteristics related to topography, visibility, forage quality and quantity, and water availability (USFWS 2000). Desert bighorn sheep prefer areas on or near mountainous terrain that are visually open, as well as steep and rocky (Wehausen 2006). Steep, rugged terrain is used for escape and lambing. Alluvial fans and washes in flatter terrain are also used for forage and water and as connectivity habitat between more rugged areas. However, based on an assessment of radiotelemetry data, Epps et al. (2007) found that desert bighorn sheep mainly used slopes greater than 10% in intermountain habitats. They used 15% slope as a cutoff value in a model for 'effective geographical distance', or EGD, where cells with slopes less than 15% were considered 10 times more costly to cross than cells with slopes greater than 15%. Because desert bighorn sheep predator avoidance is based on vigilance and visual contact, they tend to avoid dense vegetation (USFWS 2000). Peninsular bighorn sheep in particular avoid higher elevations that support chaparral.

Desert bighorn sheep occur in the following habitats (see Table 1): alpine dwarf-shrub, low sage, sagebrush, bitterbrush, pinyon-juniper, palm oasis, desert riparian, desert succulent shrub, desert scrub, subalpine conifer, perennial grassland, montane chaparral, and montane riparian (Zeiner et al. 1990). A wide range of forage

resources and vegetation associations is needed to meet annual and drought-related variations in forage quality and availability (USFWS 2000). Seasonal forage available in alluvial fans and in washes provides a diversity of browse during warmer periods that support lactation and thus is important for reproduction and recruitment of lambs. Foraging behavior is described in more detail herein.

Surface water is an important habitat element for desert bighorn sheep, although individuals can survive without drinking surface water (Wehausen 2006). While desert bighorn sheep may drink water in the cool season, in years of poor forage growth, surface water is most important during the May through October hot season, when most females and associated lambs and yearlings live largely within 2 to 3 miles of water. Males join them at these water sources as the hot season progresses with the onset of the breeding season (Wehausen, pers. comm. 2012). In populations in the eastern Mojave Desert (Old Dad Peak, Kelso Mountains, and Marl Mountains), females occur in areas closer to water and more rugged terrain than males (Bleich et al. 1997). Water sources adjacent to escape terrain are preferred and a lack of water may be a limiting factor in the distribution of desert bighorn sheep populations; there are no known large populations in regions lacking water (Wehausen 2006).

Outside the breeding season, males and females commonly occupy different habitats and usually only come together during the rut period (USFWS 2000). Females prefer particularly steep, safe areas for bearing and initial rearing of lambs (Bleich et al. 1997), especially areas of steep limestone if available (Wehausen 2006). Steep topography is not only important for lambing and rearing, but also helps desert bighorn sheep escape from predators (USFWS 2000). Because desert bighorn sheep primarily rely on their sense of sight to detect predators, open terrain with good visibility is critical for protection from predation (USFWS 2000). Males tend to occupy much less rugged habitat during the lambing season (Wehausen 2006).

Table 1. Habitat Associations for Desert Bighorn Sheep

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Alpine dwarf-shrub, Low sage, Sagebrush, Bitterbrush, Pinyon-juniper, Palm oasis, Desert riparian, Desert succulent shrub, Desert scrub, Subalpine conifer, Perennial grassland, Montane chaparral, Montane riparian,	Primary habitat	Year-round	Desert bighorn sheep prefer areas on or near mountainous terrain that are visually open and steep and rocky and that support surface water. Males tend to occupy much less rugged habitat during the lambing season.	Zeiner et al. 1990; USFWS 2000; Wehausen 2006
Alluvial fans and washes	Foraging	During warmer periods/ lambing		

Foraging Requirements

Bighorn sheep are generalist foragers and feed on a wide variety of plant species (Miller and Gaud 1989; Shackleton 1985). For example, Miller and Gaud (1989) documented 121 plant taxa in fecal samples and through direct observations of desert bighorn sheep in a Sonoran Desert habitat in Western Arizona over an 11-year period. However, the composition of their diet varies with season and location (Bleich et al. 1997; Miller and Gaud 1989; Shackleton 1985; Wehausen 2006; 74 FR 17288–17365). They must be able to access the seasonal abundance of plants at various elevations in various habitat types to maximize resources. Desert bighorn sheep adjust their feeding ranges to exploit areas with more nutritive resources, such as within bajadas, early in the season as high-protein grasses emerge. The relationship between nutritive resources, reproductive success, and optimal

timing of birth is complex. Lamb survival is strongly related to spring body growth, so the earlier they are born the more they can grow before forage quality quickly declines in late spring (Wehausen 2005). However, the earlier the birth, the more likely that ewes will have inadequate food quality during late gestation and early lactation (Wehausen 2005.) The factor that controls this relationship is the body condition of the ewes coming into the reproductive season, with ewes in better condition ovulating earlier in the season because they have the condition to withstand the period with lower nutrient resources (Wehausen 2005).

During the reproductive season, nutritious forage is typically concentrated on alluvial fans and bajadas, and in washes where more productive, wetter soils support more herbaceous forage than steeper, drier, rockier soils. These areas, therefore, are especially important food sources during the heat of summer months and in drought conditions (74 FR 17288–17365). For example, Peninsular bighorn sheep browse year-round on shrubs such as burro bush (*Ambrosia dumosa*), small-leaved hoffmannseggia (*Hoffmannseggia microphylla*), desert lavender (*Hyptis emoryi*), globemallows (*Sphaeralcea* spp.), and jojoba (*Simmondsia chinensis*). Grasses such as six weeks threeawn (*Aristida adscensionis*) and red brome (*Bromus rubens*), as well as cacti (*Opuntia* spp.), are primary food sources in the fall (74 FR 17288–17365). Forbs such as native plantains (*Plantago* spp.) and common ditaxis (*Ditaxis neomexicana*) are primary food sources in the spring (74 FR 17288–17365). The Peninsular bighorn sheep diet is about 57% shrub, 32% forbs, 8% cacti, and 2% grasses (USFWS 2000).

Desert bighorn sheep typically stay close (i.e., within 2 to 3 miles) to reliable sources of water during hot summer months and drink large quantities at each visit (USFWS 2000). Desert bighorn sheep have been known to travel at least 10 miles from perennial water sources and typically visit a water source every 2 to 3 days. Sources of water for desert bighorn sheep include rainwater accumulated in natural collection tanks and potholes in rock, natural springs, and vegetation with high water content, such as cacti (74 FR 17288–17365).

Reproduction

The primary desert bighorn breeding season, or rut period, is between August and October in the Peninsular Range (USFWS 2000) and August and November in west Mojave Desert (Wehausen 2006). The gestation period is about 6 months (range of 171 to 178 days (Shackleton et al. 1984). Desert bighorn sheep tend to have relatively high conception rates, with a reported rate of 77% to 85% (USFWS 2000). The lambing period depends on location and resources available, but generally desert bighorn sheep have a long lambing season (see Table 2 for key seasonal periods). The reported lambing period for desert bighorn sheep generally occurs between January and June, with most lambs born February to April. In the Mojave Desert, lambing occurs somewhat later than more southerly areas and may begin in December and end in June, with a small percentage of births commonly occurring in summer as well (Wehausen 2006). In a study in the Peninsular Ranges, the lambing season extended from February through August, with 87% of the lambs born from February to April (Rubin et al. 2000). Lambs usually are weaned by 6 months of age.

In the Peninsular Ranges, the reproductive age of ewes ranges from approximately 2 to 16 years of age. As the birthing time approaches, ewes seek isolated sites with shelter and unobstructed views to bear their lambs, secluding themselves from other females (USFWS 2000).

Mortality rates are highest in the first year of life and lamb survival (to 6 months of age) varies by group and year (Shackleton 1985; USFWS 2000) and is related to several factors. Reproductive success in ruminants such as desert bighorn sheep is associated with the mother's body weight, access to resources, quality of home range, and age. As discussed above, lamb survival to summer is strongly related to body growth during the spring (Wehausen 2005). Rubin et al. (2000) found that lamb survival in a Peninsular desert bighorn sheep population was related to the time of year that lambs are born, with the highest survival rate for lambs born in February through April, compared to lambs born later. Lamb mortality may also be caused by disease or disease processes complicated by environmental conditions, including habitat modification (USFWS 2000).

Winter precipitation, which is tied to plant phenology and nutrient availability for desert bighorn sheep, is an important factor in lamb survival (Wehausen 2005). In the eastern Mojave Wehausen (2005) found that rainfall in the months of October and February has the greatest effect on diet quality. Fall rainfall is important for initiating the growth of cold-tolerant species, including annuals, herbaceous perennials, and perennial grasses, and February is important for both the continued growth of cold-tolerant species, but also the growth of cold-intolerant perennial species. Timing of birthing coincides with peak nutrient availability and the amount of rainfall in the October through April period has a strong effect on lamb survival and recruitment rate (Wehausen 2005). A similar pattern was reported by Wehausen et al. (1987) for a Peninsular Range population in the Santa Rosa Mountains where rainfall in November, January and February was significantly positively correlated with lamb recruitment. Elsewhere in the desert bighorn sheep's range, similar patterns have been observed. Douglas and Leslie (1986) found a positive relationship between fall and winter precipitation and lamb recruitment the following year. Douglas and Leslie (1986) determined that 52% of the variability in lamb survival in desert bighorn population in the River Mountains in Nevada over a 12-year period was accounted for by autumn precipitation during gestation.

While precipitation patterns are strongly associated with lamb survival, lower lamb survival has also been associated periods of increased rainfall, complicating the relationship between rainfall patterns and lamb survival. Wehausen (2005) noted that declining survivorship occurs with rainfall over about 23 centimeters (about 9 inches). It has been hypothesized that increased rainfall may be associated with disease; increased standing water causes an increase in populations of *Culicoides* midges, which are a vector for bluetongue and epizootic hemorrhagic disease viruses (USFWS 2000), but Wehausen (2005) indicates that more research is needed to understand this relationship.

Table 2. Key Seasonal Periods for Desert Bighorn Sheep

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding	X	X	X	X	X	X	X	X			X	X
Lambing	X	X	X	X	X	X						X

Source: Wehausen 2006.

Spatial Behavior

Desert bighorn sheep exhibit seasonal differences in habitat use patterns (USFWS 2000), and some populations of females may migrate seasonally between mountain ranges (Jaeger 1994). Seasonal migration by desert bighorn sheep may be more common than previously thought (Wehausen, pers. comm. 2012). They tend to concentrate in areas with water during the hot summer months and expand their ranges away from water sources in the cooler, wetter season (USFWS 2000). They also alter their ranges during rutting and lambing seasons (USFWS 2000). Home range size depends on the availability of required resources, such as water, forage, and lambing habitat, and, thus, varies geographically (USFWS 2000). Forage quantity and quality, season, sex, and age also influence home range sizes. Generally, ram home ranges are larger than those of ewes. In the San Jacinto Mountains, based on a fixed kernel method for estimating home range (95% utilization distribution), the average estimated home range size was approximately 9.8 square miles for rams and 7.8 square miles for ewes (USFWS 2000).

The social structure of desert bighorn sheep is matrilineal (based on female associations). They exhibit gregarious and philopatric (remaining in natal area) behaviors (USFWS 2000). However, rams do not show the same level of philopatry as females and tend to range more widely, often moving among groups of ewes (USFWS 2000). At 2 to 4 years of age, young rams follow older rams away from their natal group during the fall breeding period, often returning after this period. Rams may use the same travel routes year after year (USFWS 2000).

Long-distance inter-mountain range dispersal movements are important for desert bighorn sheep, primarily by rams, but also by ewes (Wehausen 2006). Using radiotelemetry, Bleich et al. (1990) documented substantial intermountain movement between mountains in southeastern California. Epps et al. (2004, p. 103) state that "Three apparent natural recolonizations have been observed in recent years. It is possible that additional extinctions and subsequent recolonizations were undetected between survey years," but they do not provide more detail about these recolonizations. Nonetheless, recent information indicates that intermountain movements and natural recolonizations are not rare occurrences (Bleich et al. 1996; Epps et al. 2010). Epps et al. (2010) analyzed DNA information and found that both native and translocated desert bighorn sheep have colonized "empty habitats." Wehausen (pers. comm. 2102) reports that additional natural colonizations have occurred in several ranges, including Deep Springs, Coso, South Soda, South Bristol, Iron, Little Maria, and Cushenbury (San Bernardino Mountains). Further, ewe movements to new groups once thought be rare (e.g., USFWS 2000) are now known to be much more common (Wehausen, pers. comm. 2012). For example, 3 of 10 radio-collared females moved from the Marble Mountains to the South Bristol Mountains in 1992 when that vacant range was colonized (Wehausen, pers. comm. 2012). The available information now indicates that over the past 25 years recolonizations have exceeded the extinctions that occurred in the mid-20th Century during a 30-year drought period and during a period when desert bighorn sheep were being adversely affected by human activities (Wehausen, pers. comm. 2012).

Ecological Relationships

Access to forage and water resources in proximity to rugged escape habitat is critical for desert bighorn sheep (USFWS 2000). Because of the nutritive requirements for supporting reproduction and body growth, the quality of forage during these periods is important (e.g., USFWS 2000, Wehausen 2005). As noted previously, lambing recruitment is generally positively correlated with high winter precipitation. Poor quality forage may adversely affect maternal care if ewes are in poor condition and lamb mortality may be increased through malnutrition, thus adversely affecting recruitment (USFWS 2000). Although lack of water may adversely affect lactation, water

sources may also attract natural predators such as mountain lion (*Puma concolor*) that prey on all age classes, and coyote (*Canis latrans*) and bobcat (*Lynx rufus*) that prey on lambs (USFWS 2000). Predation may be an important loss in very small populations, including recent transplants (Zeiner et al. 1990). For this reason, it is important to have rugged escape habitat near water sources.

In addition to being sensitive to natural predators, desert bighorn sheep may be in competition with both native and non-native animals such as mule deer (*Odocoileus hemionus*), livestock, and feral burros for water and food sources (USFWS 2000). Competition with mule deer may occur in the more northern bighorn populations, but may not be as great in the Peninsular bighorn population (USFWS 2000). Cattle, sheep, and goats may be serious direct and indirect competitors for food and water sources, and may also sources of disease (USFWS 2000). Goats in particular can forage in rugged terrain favored by desert bighorn sheep and tend to overgraze, reducing or eliminating available forage for desert bighorn sheep (USFWS 2000). Cattle and desert bighorn sheep use different habitat types for grazing/browsing (Shackleton 1985), but may compete at water sites. Sheep and goats are an issue for the northern bighorn populations due to risk of disease (Wehausen 2006; Wehausen et al. 2011), but are not currently present in the Peninsular bighorn range (USFWS 2000). Present competition with cattle in the Peninsular ranges is also limited due to general absence of cattle from bighorn habitat (USFWS 2000).

Competition with cattle and feral burros in the Mojave Desert for water and food resources may occur, but a true competition between burros and desert bighorn sheep has not been demonstrated (Wehausen 2006). It is also possible that bighorn use of water sources is affected by the presence of the non-native honeybee (*Apis mellifera*) (USFWS 2000).

Domestic sheep are the major disease source for the northern bighorn populations, and sheep contact has been associated with major bighorn die-offs (Wehausen 2006). Goats also may be a disease source for desert bighorn sheep (USFWS 2000). Diseases contracted from domestic sheep and goats are described subsequently in the Threats and Environmental Stressors Section.

Population Status and Trends

Global: Subspecies *O. c. nelsoni* is apparently secure; Peninsular bighorn DPS is vulnerable (NatureServe 2010)
State: Subspecies *O. c. nelsoni* is vulnerable; Peninsular bighorn DPS is critically imperiled (NatureServe 2010)
Within Plan Area: Same as above for Peninsular bighorn DPS.

The 2009 estimate for the northern populations of Nelson's desert bighorn sheep is a population of approximately 4,800 individuals (CDFG 2010a). This compares with an estimated population of 3,737 individuals in 1972 and 4,500 individuals in 2003 (CDFG 2010a). Although the broad estimate indicates an increasing or at least stable population, local populations have shown more variability, with some local population declines (CDFG 2010a). The most recent CDFW aerial survey counts for the northern populations of the desert bighorn sheep are shown in Table 3. The large majority of the counts are within the Plan Area, with only the White Mountains Management Unit wholly outside of the Plan Area.

Table 3. Aerial Counts of Desert Bighorn Sheep in Specified Management Units for 2009–2010.

Mountain Range	Survey Date	Ewes	Lambs	Rams	Total
Management Units Within Plan Area					
Marble Mountains	October 2009	88	34	65	187
Clipper Mountains	October 2009	13	4	16	33
Kelso Peak and Old Dad Peak	October 2009	95	15	69	179
Clark, Kingston, and Mesquite Mountains	October 2009	45	6	28	79
Orocopia Mountains	September 2009	39	7	21	67
Sheephole Mountains	May 2009	22	3	17	42
South Bristol Mountains	October 2009	44	13	26	83
South Bristol Mountains	October 2010	33	9	30	72

Table 3. Aerial Counts of Desert Bighorn Sheep in Specified Management Units for 2009–2010.

Mountain Range	Survey Date	Ewes	Lambs	Rams	Total
Cady Mountains	September 2009	92	37	38	167
Cady Mountains	October 2010	102	23	49	174
San Gorgonio Wilderness Area[1]	May 2009	48	15	20	83
Subtotal Within Plan Area[2]		**485**	**116**	**315**	**916**
Management Unit Outside Plan Area					
White Mountains	March 2009	59	16	31	106
Grand Totals		**544**	**132**	**346**	**1,022**

[1] The eastern portion of the San Gorgonio Wilderness Area is within the Plan Area. The counts may include desert bighorn using areas west of the Plan Area.
[2] Subtotal excludes the 2009 counts for the South Bristol and Cady mountains to avoid double-counting.

Source: CDFG 2010a.
Note that counts are minimum population sizes because they are based on individuals actually observed during aerial surveys. Population size is assumed to be larger.

For the Peninsular bighorn sheep, as of December 2010, there were about 950 adults in nine distinct subpopulations north of the Mexican border, which indicates an upward trend since the mid-1990s (CDFG 2010a). The highest population estimate for the Peninsular bighorn was 1,170 individuals in 1974 (CDFG 2010a). Since that time, population estimates north of the Mexican border for adults have been 570 in 1988, 400 in 1992, between 327 and 524 in 1993, 347 in 1994, 276 in 1996, and 334 in 1998 (USFWS 2000).

Threats and Environmental Stressors

The potential impacts of threats and stressors are closely related to the metapopulation population structure of desert bighorn sheep in the Plan Area. Metapopulations are characterized by groups of partially isolated populations (or subpopulations) that are typically connected by emigration and immigration pathways that allow for exchange of individuals (and genetic material) and for colonizations

after local extinctions. Desert bighorn sheep exhibit such a metapopulation structure in the Plan Area in that small local populations are largely restricted to steep, isolated rocky mountain ranges that are scattered across the desert landscape and which are separated by substantial expanses of unsuitable habitat (Bleich et al. 1990; Epps et al. 2010). Based on Epps et al. (2003), there are 13 metapopulations in California, of which approximately 8 occur in the Plan Area. Within each metapopulation in the Plan Area, there are separate population groups ranging from 1 population in the San Gabriel metapopulation to 18 populations in the South Mojave metapopulation (see Table 1 in Epps et al. 2003). In the 2004 population inventory, of the most frequent population size classes in the Plan Area were either 0 or 25-100 (see Table 2 in Epps et al. 2003). As discussed in Spatial Behavior, inter-mountain movements are not rare, but conservation of the species in the Plan Area depends on maintaining intermountain habitat connectivity that allows for dispersal and migrations between populations, and recolonizations of empty habitats (Bleich et al. 1990). This intermountain habitat includes "stepping stones" within movement corridors that are not permanent habitat, but which facilitate movement (Bleich et al. 1990).

Desert bighorn sheep are threatened by loss and fragmentation of important habitats (e.g., lambing and feeding areas, escape terrain, water, travel, and dispersal routes), disease (mostly livestock derived), predation, drought, potential resource competition, and negative interactions with humans (63 FR 13136; USFWS 2000; Wehausen 2006). In addition, some of these threats are interrelated and interactive. For example, habitat fragmentation has resulted in loss of genetic diversity (Epps et al. 2005), which can result in reduced fitness and vigor and make desert bighorn sheep more vulnerable to other threat factors or stressors such as disease, drought, and predation. These kinds of threats or stressors to desert bighorn sheep are magnified in the Peninsular bighorn DPS due to reduced population numbers and consequent higher risk of extinction.

Habitat loss and fragmentation as a result of highways and aboveground canals (e.g., portions of the California aqueduct from the Colorado River to western Riverside County) and high densities of human habitation present obstacles to movement of desert bighorn sheep between mountain ranges that can interfere with the natural

metapopulation structure of desert bighorn in the Plan Area. There is essentially no migration across the Interstate highways (Wehausen, pers. comm. 2012). These physical obstacles limit the potential for natural colonization of vacant areas and gene exchange among subpopulations, which are critical to metapopulation viability (CDFG 2010a; Epps et al. 2005; Wehausen 2006). Epps et al. (2005) examined 27 separate bighorn populations in the central and southern Mojave Desert and northern Sonoran Desert had a rapid reduction in genetic diversity (up to 15%) in the 40 years or less of anthropogenic isolation. They concluded that these barriers have eliminated gene flow among populations, and that isolated populations could lose up to 40% of their pre-isolation genetic diversity over the next 60 years.

Historically, disease contracted from domestic sheep has probably been the greatest factor in desert bighorn sheep population declines throughout its range in North America (USFWS 2000; Wehausen 2006). Extensive domestic sheep grazing in northeastern California, northern Nevada, southwestern Idaho, Oregon, and Washington, likely lead to the extirpation of all native populations in these regions. In contrast, where domestic sheep grazing has not been economical, such as Canada and Alaska, little change has occurred in the distribution of native sheep (Wehausen 2006).

Wehausen et al. (2011) provide a comprehensive review of experimental research on the risk of respiratory disease transmission from domestic sheep to bighorn sheep (the so-called "contact hypothesis), including (1) contact trials between bighorn sheep, domestic sheep and other native and domestic animals; (2) inoculation experiments with no animal contact; (3) studies to isolate and identify specific organism (i.e., bacterial strains and other pathogens) that may be responsible for pneumonia in bighorn sheep; and (4) vaccination experiments. Their review found that the experimental evidence supports the contact hypothesis. Contact between domestic sheep and bighorn sheep, as well as inoculation with certain strains of the bacteria *Mannheimia haemolytica* cultured from the respiratory tracts of domestic sheep, has a high probability of causing fatal pneumonia in the bighorn sheep. At least one study also found that *Pasturella multicoda* cultured from a flock of wild and domestic sheep cause fatal pneumonia in bighorn sheep (Callan et al.

1991). As a test of the domestic sheep-bighorn sheep contact hypothesis, contact trials between bighorn sheep and other native and domestic animals produce low disease and mortality rates, indicating that the high disease and mortality rates of bighorn sheep in contact with domestic sheep are not an artifact of captivity (which was an alternative hypothesis) (Wehausen et al. 2011). The studies of specific organisms responsible for pneumonia in bighorn sheep after contact with domestic sheep failed to clearly identify specific causes (possibly due to the complexity of the disease and/or the sensitivity of culturing methods in identifying the sampled microbial community); nonetheless, the research has clearly demonstrated a negative effect of direct contact between bighorn sheep and domestic sheep despite uncertainty of the nature of the pathogen. Finally, vaccinations failed to reduce the spread of respiratory disease and vaccination is probably not an effective management tool, both because it apparent lack of effectiveness and the logistical challenges in treating wild populations (Wehausen et al. 2011).

Predation is also a significant factor in desert bighorn sheep mortality, with mountain lion being the major predator. In the Kingston, Clark, and Granite mountains, considerable predation by mountain lion has been documented (Jaeger 1994; Wehausen 1996). In the Granite Mountains, mountain lion predation caused a steep population decline in the desert bighorn sheep population, with the population reduced to 8 ewes for a period of 3 years (Wehausen 1996). In this study all mortalities in the first 3 years of the study were from mountain lion predation (Wehausen 1996). Predation abated after the first 3 years of the study and the population rebounded at 15% annually the next 3 years (Wehausen 1996). Areas of the Mojave Desert where mountain lion predation is a threat to desert bighorn sheep also support populations of native or introduced deer, which is the mountain lion's primary prey (Wehausen 2006). At least four radio-collared male desert bighorn sheep in the eastern Mojave Desert were killed by mountain lions; predation of females was not confirmed and only males tended to use habitats with mountain lions (Bleich et al. 1997). In the Peninsular Ranges, predation is also a frequent cause of mortality. Of 61 documented mortalities of radio-collared sheep from 1992 to 1998 between Highway 74 in the Santa Rosa Mountains and the Mexican border, 42 were attributed to mountain lion (USFWS 2000). Another

study of mortality conducted from 1991 to 1996 in the northern Santa Rosa Mountains found that predation accounted for 9 of 32 adult desert bighorn sheep mortalities, of which, 8 were due to mountain lion predation and 1 due to either mountain lion or bobcat predation (USFWS 2000). Coyote and bobcat also prey on desert bighorn sheep, but are more likely to take lambs; a study showed that of nine lamb mortalities recorded in 1998 and 1999, five were attributed to coyote or bobcat predation (USFWS 2000).

Prolonged drought periods can also cause population declines (USFWS 2000; Wehausen 2006). As discussed previously, high-quality forage associated with winter precipitation and water sources are important to support reproduction (e.g., USFWS 2000; Wehausen 2005, 2006). Lamb recruitment is reduced during periods of drought because gestation or lactation is disrupted or maternal care by ewes in poor condition is reduced, leaving the lambs vulnerable to malnutrition and predation. Drought can increase competition with native and non-native species, such as livestock, for food and water sources (Wehausen 2006). Competition for water sources can also increase congregations around water, thus increasing the risk of disease transmission (USFWS 2000). Epps et al. (2004) examined whether local extinctions of historical desert bighorn sheep populations are correlated with regional climate patterns and found that elevation, precipitation, and availability of dependable springs are strongly related to population persistence. They concluded that climate has already affected local extinction patterns and that desert bighorn sheep are vulnerable to the effects of future climate change, especially if precipitation is reduced in association with climate change. However, while observations of local extinctions are consistent with directional climate change, Epps et al. (2004) also noted that natural climate stochasticity cannot be ruled out as a factor, with population expansions during cooler wetter periods and retreats during periods of increase drought frequency and intensity. It is unknown long-term climate change is the cause of current population trends (Epps et al. 2004).

Within the Peninsular Ranges, negative interactions with humans and pets, and other urban-related factors, are a threat to the Peninsular bighorn sheep (USFWS 2000). In addition to loss and fragmentation of habitat due to urban and rural development, more than 30% of

mortalities in one study were directly attributable to human activities, including vehicle collisions, poisoning, and entanglement in fences (USFWS 2000). Humans, pets, off-road vehicles, construction activities, and aircraft also can affect desert bighorn sheep behavior (Leslie and Douglas 1980; USFWS 2000). These factors can affect desert bighorn sheep to the extent that essential activities, such as foraging or the use of important areas (e.g., water sources, mineral licks, lambing areas, traditional movement routes), are disrupted, which can affect the viability of populations through reduced lamb recruitment (USFWS 2000). Human activities may also induce physiological stress such as increased heart rate, which can affect the health of desert bighorn sheep individuals and lamb recruitment (USFWS 2000). Impacts related to human activities may also occur in the northern populations. However, with the exception of livestock grazing and some recreational activities, impacts would be expected to be less frequent or severe due to reduced human activity in the more remote areas occupied by desert bighorn sheep.

Non-native plants used for landscaping, such as oleander (*Nerium oleander*) and laurel cherry (*Prunus laurocerasus*), have been implicated in the poisoning of desert bighorn sheep (USFWS 2000). Tamarisk (*Tamarix* spp.) is highly consumptive of water, reducing critical surface water sources for desert bighorn sheep (USFWS 2000).

Mortality in a desert bighorn sheep population in the vicinity of Old Dad Peak was linked to type C botulinum (*Clostridium botulinum*) poisoning near two artificial water catchments (guzzlers) (Swift et al. 2000). The investigators reconstructed the probable cause of the poisoning as 13 lambs that fell into and drowned in one guzzler tank while attempting to drink from the top of the tank. A hatch cover had become dislodged when the drinker trough was dry because the tank valve was closed. The decaying lamb carcasses served as the substrate for the growth of *Clostridium botulinum*, which other individuals ingested after a rain increased water levels and allowed sheep to drink from the source (Swift et al. 2000).

Conservation and Management Activities

The Bureau of Land Management (BLM), CDFG, state parks, National Park Service, and private non-profit organizations (the Bighorn

Institute, the Anza-Borrego Foundation, Society for the Conservation of Bighorn Sheep, and Desert Wildlife Unlimited, Inc.) have planned implemented and/or participated in numerous conservation and management actions that benefit the desert bighorn sheep.

Conservation and management activities undertaken by the BLM to benefit the Peninsular desert bighorn sheep include the following actions identified in the Recovery Plan (USFWS 2000):

- Installation of gap fencing to eliminate cattle grazing from steep terrain and from water sources in canyons
- Reduction in grazing pressure on allotments
- Closure of most routes of travel east of McCain Valley Road, except to private inholdings, to ranchers, and to Carrizo and Sacatone overlooks
- Designation of wilderness study areas and subsequent management for non-impairment of wilderness values
- Designation of Jacumba, Carrizo Gorge, Coyote Mountains, Sawtooth Mountains, Fish Creek Mountains, and Santa Rosa wilderness areas by Congress, with attendant elimination of vehicular access
- Tamarisk control efforts around water sources
- Establishment of the Santa Rosa Mountains National Scenic Area Visitors Center to provide public education
- Financial assistance to the Bighorn Institute during its formative years, as well as land transfer and lease under the Recreation and Public Purposes Act
- Temporary closure to dogs on most lands in the Santa Rosa Mountains National Scenic Area
- Closure of roads into Dead Indian Canyon and Carrizo Canyon
- Designation of Santa Rosa and San Jacinto Mountains National Monument, which will prohibit mining and off-road vehicle use on federal lands, support coordinated land management by federal agencies, and increase the area's funding priority.

The BLM also issued an Instruction Memorandum in 1992 regarding domestic sheep grazing, such that domestic sheep should not be allowed within 9 miles of desert bighorn habitat, except where topographic features or other barriers prevent physical contact. Also, domestic sheep trailed and grazed outside the 9-mile zone in the vicinity of desert bighorn sheep habitat should be closely managed and carefully herded (Wehausen 2006).

CDFG manages desert bighorn sheep populations throughout much of the state through the Desert Bighorn Sheep Conservation Program (CDFG 2010a). In accordance with Section 1801 of the California Fish and Game Code, the state policy is to preserve, restore, utilize, and manage the desert bighorn sheep population. Limited harvest of desert bighorn sheep (excluding the Peninsular DPS and the Sierra Nevada bighorn sheep (*O. c. sierra*) which are fully protected) in selected areas is provided by state law for biologically sound management (CDFG 2010a). Management of desert bighorn sheep includes sport hunting of rams, with a limit on hunting tags for no more than 15% of the ram population in a single year (CDFG 2010a). As part of the management program, CDFG is required to report the status of management units; summarize counts of individuals in specified management units (see Table 3); report the number of hunting tags issued; summarize unlawful take of desert bighorn sheep; report the number of individuals translocated; and track the environmental impacts of hunting (CDFG 2010a).

CDFG conducts periodic inventories of the distribution of desert bighorn sheep in California in specific management units to assess population trends and provide the basis for issuance of hunting tags (see Table 3 for the 2009–2010 counts).

CDFG has also prepared management plans for a number of the major herds in California. The CDFG Desert Bighorn Sheep Management Program is currently preparing a range-wide management program that will provide a strategy to conserve populations throughout the state (CDFG 2010a). In 2010, draft regional management plans were prepared and submitted for approval for the Cady Mountains and South Bristol Mountains management units (CDFG 2010a). These plans address the following issues (CDFG 2010b, 2010c):

1. The numbers, age, sex ratios, and distribution of desert bighorn sheep within the management unit

2. Range conditions and a report on the competition that may exist as a result of human, livestock, wild burro, or any other mammal encroachment
3. The need to relocate or reestablish bighorn populations
4. The prevalence of disease or parasites within the population
5. Recommendations for achieving the policy objective of Section 4900, which addresses the potential for limited hunting opportunities for desert bighorn sheep.

A management objective of the state conservation program is to re-establish desert bighorn sheep on historical ranges (CDFG 2010a). Since 1983, CDFG has translocated almost 500 individuals (including the Sierra Nevada subspecies *O. c. sierrae*).

CDFG also conducts capture-sample-radio collar-release studies for research purposes. In 2010, 10 individuals were captured-collared-released in the Santa Rosa and Vallecito mountains, including 9 ewes and 1 ram (CDFG 2010a).

Anza-Borrego Desert State Park supports a majority of the range-wide Peninsular bighorn sheep population in California. Anza-Borrego Desert State Park has been actively involved in the conservation of Peninsular bighorn sheep for 30 years. Specific activities relevant to the DRECP that were identified in the Recovery Plan (USFWS 2000) are as follows:

- Construction of guzzlers to supplement water supplies
- Annual monitoring (conducted for 40 consecutive years; California Department of Parks and Recreation 2009)
- Research into bighorn sheep ecology and threats
- Tamarisk removal from riparian areas within bighorn sheep habitat to enhance water availability and native plant community regeneration (approximately 120 miles of canyons and stream courses had been treated by 2000)
- Seasonal access closure of bighorn sheep watering areas from June 1 to October 1
- Remove feral cattle from bighorn sheep habitat

- Construct gap fencing to keep stray cattle from entering bighorn sheep habitat
- Public outreach, including production of a 15-minute movie "The Bighorn of Anza-Borrego"
- Closure of some areas to vehicular traffic.

The National Park Service has conducted burro removal from their lands in the Mojave Desert, with the goal of removing all approximately 1,300 burros from the Mojave National Preserve between 1998 and 2001. (http://www.nature.nps.gov/yearinreview/yir98/chapter06/chapter06pg2.html). Although true competition between desert bighorn sheep and burros has not been demonstrated (Wehausen 2006), burros have caused adverse impacts on native plant communities, wildlife, soils, water quality (http://www.nature.nps.gov/yearinreview/yir98/chapter06/chapter06pg2.html).

The Bighorn Institute is a nonprofit organization formed in 1982 that investigates the causes of desert bighorn sheep declines, particularly among Peninsular bighorn sheep. The institute began monitoring radio-collared desert bighorn sheep in the northern Santa Rosa Mountains in 1982 and the San Jacinto Mountains in 1992. Research activities conducted by the institute include the ecology of bighorn populations in the Santa Rosa and San Jacinto mountains, lamb ecology, captive breeding and wild population augments, annual population surveys, and disease research (Bighorn Institute 2011).

The Anza-Borrego Foundation is the nonprofit cooperating association for the Anza-Borrego Desert State Park and is a sponsor for the annual desert bighorn sheep count, which has been conducted from 1971 through 2010.

The Society for Conservation of the Bighorn Sheep (SCBS) is a nonprofit organization established in 1964 that has several programs for restoring desert bighorn sheep (http://sheepsociety.com/) in coordination with CDFG and BLM. The SCBS provides labor to help conduct censuses and to establish "drinker" sites and also conducts water monitoring (including remote water monitoring stations that record available water at drinkers and precipitation) and water

hauling to supplement water at some sites. SCBS maintains remote trail cameras to monitor wildlife use of water sites. SCBS also has "Area Captains" that volunteer under the auspice of CDFG and conduct inspections of the drinkers twice a year and "Hot Shot Crews" that conduct repair and maintenance at drinkers.

Desert Wildlife Unlimited, Inc. is also involved in providing and maintaining Drinkers for desert wildlife, including desert bighorn sheep (http://www.desertwildlifeunlimited.com/home/). They employ 12,000 gallon fiberglass tanks with a step drinker attached, which require relatively little maintenance.

Data Characterization

Data availability for desert bighorn sheep is excellent and represents one of the best population datasets for any managed species in California. In particular, the Peninsular bighorn sheep DPS has been monitored annually since 1971. Furthermore, extensive research on the ecology of the desert bighorn sheep has yielded an excellent understanding of its habitat and ecological relationships.

The CDFG, State Parks, Anza-Borrego Foundation, and the Bighorn Institute conduct periodic assessments of the desert bighorn sheep populations in California, including portions of the Peninsular bighorn DPS. CDFG assessments are based on historical and current data from ground, waterhole, and aerial surveys that are suitable for estimating population size classes (CDFG 2010a). The Bighorn Institute conducts annual assessments of bighorn populations in the Northern Santa Rosa and San Jacinto mountains, and includes radiotemeletry data to study habitat use, reproduction, survival, mortality, and general ecology (Bighorn Institute 2011). The annual desert bighorn sheep count in Anza-Borrego Desert State Park has been conducted annually since 1971 and includes mid-summer counts of ewes, lambs, male and female yearlings, and rams in about 21 different locations in the park (California Department of Parks and Recreation 2009).

Management and Monitoring Considerations

The CDFG (2010b, 2010c) identified several management and monitoring considerations for desert bighorn sheep, including

demography (numbers, age, sex ratios, and distribution of desert bighorn sheep within management units); range conditions; relocation or reestablishment of populations; and the prevalence of disease or parasites.

The BLM West Mojave Plan determined that the best way to ensure the long-term viability of desert bighorn sheep metapopulations would be by preventing further population losses and fragmentation and restoring populations in vacant historical habitat. Natural and induced colonization may require artificial enhancement of populations, such as water developments (Wehausen 2006). Contact between domestic sheep and desert bighorn sheep should be prevented by eliminating or carefully managing sheep grazing in the vicinity of desert bighorn sheep habitat (Wehausen 2006). To ensure reliable water supply during the summer months, key water sources within current and historical desert bighorn sheep habitat should be closely monitored and potentially enhanced. Water enhancement may promote development of large desert bighorn sheep populations that may produce natural colonists to reestablish populations in vacant habitat (Wehausen 2006). However, because water sources may also enhance the populations of desert bighorn sheep predators, such as mountain lion, coyote, and bobcat, water enhancement should be limited.

The federal *Recovery Plan for Bighorn Sheep in the Peninsular Ranges, California* (USFWS 2000) identified improving adult survivorship as likely the strongest positive influence on Peninsular bighorn population dynamics in the short term. Over the long term, conservation and effective management of conserved lands are needed to recover the Peninsular bighorn sheep. Minimizing adverse effects of human disturbance by preventing further fragmentation is critical to the persistence of ewe groups bordering the Coachella Valley. Maintaining adequate buffers between urban development and Peninsular bighorn sheep habitat, and effective management of human activities within ewe group home ranges is needed (USFWS 2000).

Habitat fragmentation and population isolation has led to decreased genetic diversity in small isolated populations (Epps et al. 2005). Fragmentation of metapopulations from fenced highways, aqueducts, and losses of some populations should not be permitted. Epps et al.

(2005) recommend that existing barriers to movement should be mitigated and new highways in desert bighorn sheep habitat should be designed to minimize disruption of connectivity. Fencing near existing drainage undercrossings should be modified to allow access to the undercrossings and construction of overpasses should be considered to reestablish connectivity (Epps et al. 2005).

When reintroduction stock is available, historical habitat should be restocked to maximize connectivity and the number of populations in remaining metapopulations. Although evidence suggests that existing metapopulations can remain viable if adequately managed and intermountain travel corridors are maintained, opportunities to reestablish connections across recent artificial barriers that now define metapopulations should be considered (Wehausen 2006).

Species Modeled Habitat Distribution

The habitat model used for the Plan Area was provided by BLM and depicts mountain ranges and intermountain habitat for desert bighorn sheep suitable for both supporting local populations (i.e., mountain habitat) and movement (i.e., intermountain habitat). There are 12,872,136 acres of modeled suitable habitat for desert bighorn sheep in the Plan Area, including 7,976,800 acres of mountain habitat and 4,893,423 acres of intermountain habitat.

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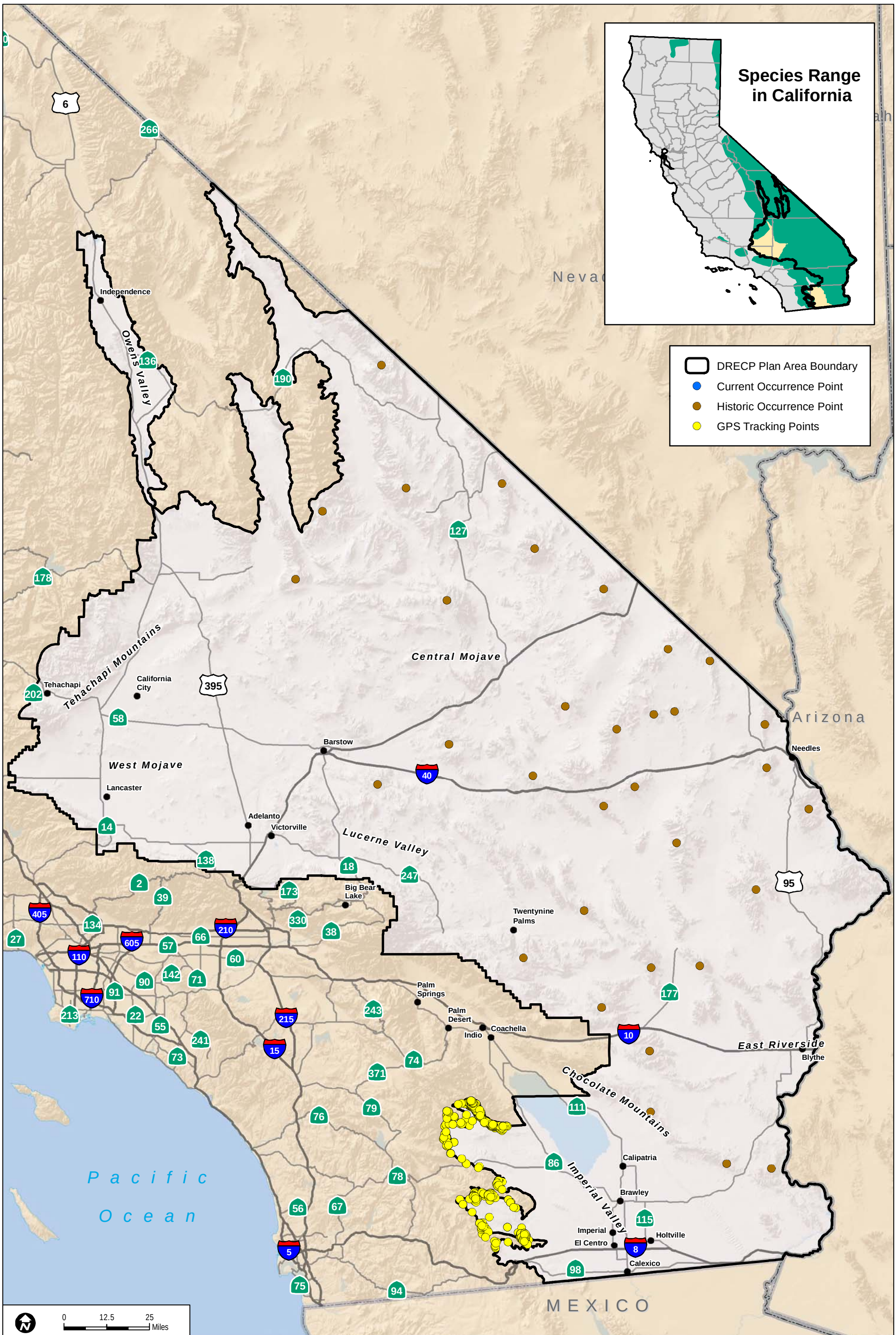

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CDFW (2013); CWHR (2008)

FIGURE SP-M01

Bighorn Sheep Species Occurrences in the Plan Area

Desert Kit Fox (*Vulpes macrotis arsipus*)

Legal Status

State: None
Federal: None
Critical Habitat: N/A
Recovery Planning: N/A
Notes: There is disagreement about the taxonomic relationship of kit fox (*Vulpes macrotis*) and swift fox (*V. velox*) and subspecific designations for kit fox (e.g., Dragoo et al. 1990; Mercure et al. 1993; 57 FR 28167–28169).

Taxonomy

The kit fox (*V. macrotis*) is in the family Canidae and is the smallest canid species in North America (McGrew 1979). Descriptions of its physical characteristics can be found in McGrew (1979). While the desert kit fox (*V. m. arsipus*) is referred to in this profile as a subspecies of the kit fox, the taxonomy of this group has been uncertain and controversial, both at the species and subspecies levels. Dragoo et al. (1990) concluded that, based on genetic data, all arid-land foxes in North America pertained to a single species, the swift fox (*Vulpes velox*), and that morphometric data indicated that all kit foxes should be recognized as a subspecies of the swift fox. However, in a 90-day finding regarding a petition to remove the federally listed endangered San Joaquin kit fox (*V. macrotis mutica*) subspecies from the endangered species list based on the argument that the subspecies was not a valid taxon, the U.S. Fish and Wildlife Service (USFWS) asserted that the morphometric data presented by Dragoo et al. (1990) acknowledged the separation between the kit fox and swift fox (57 FR 28167–28169). The USFWS further cited a yet unpublished genetic study indicating that the mitochondrial DNA haplotype of the kit foxes and swift foxes was more geographically structured than that of larger canids and that gene flow between the two taxa was restricted (57 FR 28167–28169). The results of the genetic study cited in the 90-day finding were later published by Mercure et al. (1993),

which supported the conclusion that kit fox and swift fox were separate species. However, Mercure et al. (1993) also concluded, with the exception of the San Joaquin kit fox, that the genetic data did not support the other 10 subspecific designations of kit fox, including desert kit fox. Currently, no subspecies of kit fox are recognized, including desert kit fox and San Joaquin kit fox (Wilson and Reeder 2005). However, Mercure et al. (1993) acknowledged that the Colorado River may be a barrier to gene flow and that more extensive sampling would be needed to understand microgeographic barriers to gene flow such as the Colorado River.

Given that the desert kit fox subspecies is not listed as threatened or endangered, or otherwise has special state or federal status, these taxonomic issues are not relevant to its status as Covered Species under the Desert Renewable Energy Conservation Plan (DRECP).

Distribution

General

For the purpose of this profile, the range of the desert kit fox (*V. m. arsipus*) as described by Hall (1981) for *V. velox arsipus* is used. The desert kit fox is a year-round resident of the southwestern deserts of California, southern Nevada, the lower elevations of western and southern Arizona, and northern Mexico. Its western boundary that separates it from the federally listed and isolated San Joaquin kit fox subspecies is the Antelope valley in the west Mojave. The Tehachapi and Southern Sierra Mountain ranges form a physical barrier between desert kit fox and San Joaquin kit fox, although Mercure et al. (1993) suggest that the lower elevation Tehachapi range may be more permeable to movement than the Southern Sierra range.

Distribution and Occurrences within the Plan Area

Historical

The desert kit fox's range historically included the entire Plan Area.

Recent

There is a general lack of recent distribution information for this species; however, the desert kit fox's current distribution is considered to include the entire Plan Area.

Natural History

Habitat Requirements

Kit foxes generally inhabit arid regions that receive less than about 16 inches (400 millimeters) of rain annually (Tannerfeldt et al. 2003). In the Plan Area, desert kit fox primarily occurs in open desert scrub habitats on gentle slopes. Creosote bush scrub in California is the most common habitat association for desert kit fox in California (McGrew 1979). A similar association with creosote brush scrub for den sites has been documented in Arizona (Zoellick 1985; Zoellick et al. 1989). In the Great Basin Desert portion of the Plan Area, suitable habitat includes saltbush (*Atriplex* spp.) scrubs. Penrod et al. (2012) created a suitable habitat model for desert kit fox that covers the Plan Area and that incorporates vegetation, topography, and road density and classifies habitat as good, fair, marginal, and unsuitable. "Good" habitat includes creosote bush–white bursage desert scrub or mixed salt desert scrub on slopes less than 5% and with low road density. "Fair" habitat includes areas with slopes less than 5% and other vegetation types suitable for kit fox such as playas and washes or medium road densities. "Marginal" habitat includes areas with slopes of 5%–15% or vegetation/cover types marginal for kit fox such as dune fields. "Unsuitable" areas includes slopes greater than 15%, unsuitable vegetation/cover types such as unvegetated lands, rocklands, bedrock, cliff and outcrop, and developed and cultivated lands.

O'Farrell and Gilbertson (1986) documented desert kit foxes in the western Mojave Desert northeast of California City and south of the El Paso Mountains (Rand Open Area and Desert Tortoise Research Natural Area) using habitat dominated by *Larrea*-S*chismus*-*Erodium*, with relatively low cover of burro bush (*Ambrosia dumosa*). O'Farrell and Gilbertson (1986) characterized the study sites as disturbed by sheep grazing and off-highway vehicles (OHVs). Similarly, kit foxes in western Arizona were observed to den in creosote scrub and spend

more time in creosote scrub than expected based on its availability relative to other habitat types (Zoellick et al. 1989). About 80% of kit fox dens in the Great Basin Desert in western Utah were in sparsely vegetated shadscale flats with low vegetation of 8–10 inches, and with shadscale (*Atriplex confertifolia*) as the most common species (Egoscue 1956). Egoscue (1956) noted that while dens were located in areas with low vegetation and high visibility, prey productivity was low in these areas, requiring individuals to travel more than a mile to forage in more productive habitats. However, Arjo et al. (2003) discuss a potential tradeoff of vegetation structure around dens, with lower vegetation height providing better detection of advancing predators and higher vegetation height providing better concealment and possibly higher invertebrate prey availability. Proximity of water does not appear to be a factor in kit fox den selection (Egoscue 1956), and the species can meet it water needs through prey (McGrew 1979).

Dens are an important resource for kit fox because they provide microclimate moderation and protection from predators, and may be a limiting resource for kit fox distribution (Arjo et al. 2003). Kit foxes form monogamous pairs (at least through a breeding season) and often small family groups that occupy den complexes (Ralls and White 2003; Ralls et al. 2007). Kit foxes may dig their own dens, use dens created by other species such as badger (*Taxidea taxus*), or expand on burrows created by smaller species such kangaroo rats (*Dipodomys* spp.) and prairie dogs (*Cynomys* spp.) (Arjo et al. 2003; Tannerfeldt et al. 2003). Whether kit foxes dig their own dens or use dens and burrows created by other species may depend on the availability of preexisting dens/burrows, with kit foxes rarely digging dens when they do not have to (Tannerfeldt et al. 2003). Desert kit fox dens in the western Mojave in the O'Farrell and Gilbertson (1986) study tended to be on west- and northwest-facing slopes on friable soils with an absence of stones, caliche, or hardpan (O'Farrell and Gilbertson 1986). Kit foxes may also occasionally den in manmade culverts (Egoscue 1956; O'Farrell and Gilbertson 1986). Arjo et al. (2003) discuss the hypothesis that the orientation of natal den entrances may be related to protection from prevailing winds and provide other microclimatic advantages, suggesting that entrance orientation may be related to local climatic factors. Selection of den sites may also depend on the distribution of coyotes (*Canis latrans*), which is a common natural

predator of kit foxes (e.g., Rall and White 1995; White et al. 1995; White and Garrott 1997; Kozlowski et al. 2008) and direct competitor for resources (White et al. 1995; Arjo et al. 2003, 2007; Kozlowski et al. 2008). For example, in western Utah, kit foxes may have altered their distribution and den sites to more mountainous areas and areas vegetated by non-native grasses in response to increased coyote populations in the study area since 1959 (Arjo et al. 2003) (see discussion in Ecological Relationships).

Kit fox dens typically have multiple entrances (Egoscue 1956; O'Farrell and Gilbertson 1986; Tannerfeldt et al. 2003). In the O'Farrell and Gilbertson (1986) study, dens averaged 3–5 entrances, with up to 10 entrances. Natal (pupping) dens used by desert kit foxes from January to the end of May were larger and had more entrances (5–8) than non-natal dens (3–4) used from June through December (O'Farrell and Gilbertson 1986), which also appears generally common in kit foxes (e.g., Arjo et al. 2003; Tannerfeldt et al. 2003).

Kit foxes use numerous dens, switching dens frequently, and dens tend to be clustered (Tannerfeldt et al. 2003). Clusters include several dens (in one study, up to 17) that may be more than 328 feet (100 meters) apart (Tannerfeldt et al. 2003). In San Joaquin kit fox, den switching may occur several times monthly and most often during the dispersal season, but switching is also related to age class with adults tending to use more dens than juveniles (Tannerfeldt et al. 2003). Although dens may be shared by pair-mates throughout the year, den sharing may be seasonally variable, with higher rates during December during the breeding season and lower rates in February when very young pups were present, for example (Ralls et al. 2007).

Natal dens in the western Mojave appeared to be spaced, with possible territorial exclusivity, with a minimum inter-den distance of approximately 1.25 miles (2 kilometers) (O'Farrell and Gilbertson 1986). This spacing may reflect territorial requirements and carrying capacity (O'Farrell and Gilbertson 1986). Similarly, in western Utah natal dens were at least 2 miles (3.2 kilometers) apart (Egoscue 1975). In San Joaquin kit fox, territories of adjacent social groups had only slight overlap (White and Ralls 1993).

Selection of den sites does not appear to be strongly related to nearby human activities, nor do kit foxes appear to actively avoid man-made features such as roads and structures. O'Farrell and Gilbertson (1986) found that most desert kit fox dens were within 492–656 feet (150–200 meters) of roads or trails in the western Mojave. Bjurlin et al. (2005) found that almost 10% of San Joaquin kit dens in the Bakersfield area were within 100 feet of road centerlines and that some dens used features of major roads, including culverts, embankments and underpasses, and drainage basins or canals immediately adjacent to roads.

Foraging Requirements

Several studies in California, Arizona, and Utah, as summarized by Tannerfeldt et al. (2003), show that the primary food sources for kit foxes are rodents and lagomorphs, including jackrabbit (*Lepus* spp.) and cottontails (*Sylvilagus* spp.). Egoscue (1956) listed several prey species in the Great Basin Desert of western Utah, including black-tailed jackrabbit (*Lepus californicus*), kangaroo rat (*Dipodomys* spp.), and deer mouse (*Peromyscus maniculatus*), but also burrowing owl (*Athene cunicularia*), western meadowlark (*Sturnella neglecta*), horned lark (*Eremophila alpetris*) (notably all open ground-nesting species), side-blotched lizard (*Uta stansburiana*), and sand cricket (*Stenopelmatus* sp.). Similarly, on the Carrizo Plain in California, San Joaquin kit fox prey included kangaroo rats, pocket mice (*Chaetodipus* spp. and *Perognathus* spp.), deer mouse., black-tailed jackrabbit, desert cottontail (*Sylvilagus audubonii*), and California ground squirrel (*Spermophilus beecheyi*) (White and Ralls 1993). In the Plan Area, it is expected that primary prey for desert kit fox include black-tailed jackrabbit, desert cottontail, Merriam's kangaroo rat (*D. merriami*) (the most common and widespread kangaroo rat in the Plan Area), various pocket mice species, other rodents such as woodrats (*Neotoma* spp.) and California ground squirrel, and various small reptiles.

Hunting is almost strictly nocturnal, with kit foxes resting in their dens during the day (Egoscue 1956; White et al. 1995). As noted under spatial activity, individuals may move several miles daily, but it is likely that foraging distances are closely related to prey availability, which is likely variable spatially and temporally (Egoscue 1956).

Reproduction

The desert kit fox reproductive period in the Plan Area is generally December to late May (O'Farrell and Gilbertson 1986) (see Table 1), which is consistent with other parts of the kit fox's range (e.g., Egoscue 1956; McGrew 1979). In the O'Farrell and Gilbertson (1986) study in the western Mojave, males maintained scrotal development throughout the year, but females were reproductive in December and January. Gestation is approximately 49–56 days (McGrew 1979), and females in the O'Farrell and Gilbertson (1986) study were lactating in March and April, indicating birth in February and March. Kit fox litters are 2–6 pups (Egoscue 1956; McGrew 1979; Tannerfeldt et al. 2003; USFWS 2010), and pups emerge from the natal den at about 4 weeks of age (USFWS 2010). Both adults provide care to pups. Initially males do most of the hunting while lactating females remain in the den (Egoscue 1956). In the O'Farrell and Gilbertson (1986) study, pups were absent from natal dens by the end of May. However, for San Joaquin kit fox, pups remain under the care of adults for 4 to 5 months, before beginning to disperse from their natal area as early as July and continuing through August and September (Moonjian 2007; USFWS 2010). Some offspring remain with their parents and help raise the next litter during the following year (USFWS 2010). Also in San Joaquin kit fox, yearling females may breed, with about 18% of monitored successfully reproducing (Cypher et al. 2000). Egoscue (1956) reported two lactating females in the same den on two occasions, with one instance apparently a mother and daughter.

Kit foxes generally exhibit monogamy, with pairs remaining together for several breeding seasons, and some pair bonds being permanent until the death of one of the pair (Egoscue 1956; O'Farrell and Gilbertson 1986; Ralls et al. 2007). In San Joaquin kit fox, Ralls et al. (2007) documented that 14 of 16 dissolutions of a pair were due to the death of a pair-mate, 1 was due to the male abandoning the female, and the other was due to a new male displacing the mate. Pair formation can occur throughout the year (Ralls et al. 2007).

Mortality rates in the O'Farrell and Gilbertson (1986) study were high with average observed longevity on the order of 10 months (range 8–14), although some individuals were still alive when the study was completed. Mortality resulted from several causes, including shooting,

starvation, predation (likely coyote or dog), vehicle collisions, and den collapse (see Threats and Environmental Stressors). In a study of dispersal by San Joaquin kit fox, Koopman et al. (2000) found that more than 65% of dispersing juveniles died within 10 days of leaving their natal range. The primary cause of mortality of dispersing and philopatric juveniles was predation. Kit foxes in zoos have lived 10–12 years (McGrew 1979), but such a long life span in the wild is unlikely.

Table 1. Key Seasonal Periods for Desert Kit Fox

	Jan	Feb	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding	x	x	x	x							x
Dispersal						x	x	x			

Spatial Activity

Desert kit foxes are quite mobile and have relatively large home ranges. In the western Mojave, O'Farrell and Gilbertson (1986) estimated ranges of the approximately 494 acres based on radiotelemetry data. Data for other subspecies indicate at least as large to much larger home ranges, with home-range size likely related to resource availability. For San Joaquin kit fox, Koopman et al. (2001) determined a mean adult home-range size of approximately 1,072 acres and a mean pup home-range size of 325 acres on the Naval Petroleum Reserves in western Kern County (USFWS 2010). Briden et al. (1992, as cited in USFWS 2010) found that denning ranges (the area encompassing all known dens for an individual) for San Joaquin kit fox averaged approximately 1,169 acres in western Merced County. White and Ralls (1993) estimated a mean home range for San Joaquin kit fox of approximately 2,866 acres at the Carrizo Plain in 1990 and 1991, but noted these home ranges were large and likely reflected drought conditions and prey scarcity. Home ranges during this study were also relatively exclusive, with little overlap between individuals of the same sex (White and Ralls 1993). At the Camp Roberts Army National Guard Training Site in northern San Luis Obispo County, radiotelemetry documented mean home ranges for San Joaquin kit fox of approximately 5,782 acres (Root and Eliason 2001, as cited in USFWS

2010). White and Ralls (1993) suggested that large, exclusive home ranges during periods of drought may be an adaptation to episodic prey scarcity and a means to maintain their own body mass and condition.

Daily movements of desert kit foxes in western Arizona during the period of December through March averaged 8.9 miles (14.3 ±0.71 kilometers/night) for males and 7.4 miles (11.8 ±1.08 kilometers/night) for females (Zoellick et al. 1989). Males tended to move greater distances during the breeding season compared to pup rearing and pair formation periods (Zoellick et al. 1989). O'Farrell and Gilbertson (1986) did not observe young remaining in their natal territory and recorded a maximum dispersal of approximately 20 miles (32 kilometers) by a female. Egoscue (1956) reported movements up to 20 miles by juvenile kit foxes in western Utah. However, in the San Joaquin kit fox, which has been much more extensively studied than desert kit fox in the Plan Area, some offspring remain with their parents (Ralls et al 2001). Young of this subspecies may also remain their natal territory. In one study spanning 16 years, 33% of tracked juveniles dispersed from their natal territory, with significantly more males dispersing than females, and the average dispersal distance was 4.8 miles (range of 1.1 to 20 miles) (Koopman et al. 2000). Most dispersal occurred in the first year of the animal's life. Briden et al. (1992, as cited in USFWS 2010) documented dispersals of 1.2 to 12 miles. Four long-distance dispersals of between 25 and 50 miles were documented between Camp Roberts and Fort Hunter Liggett Military Reserve in Monterey County and the Carrizo Plain (California Air National Guard 2008, as cited in USFWS 2010).

Koopman et al. (2000) did not find any significant relationships between dispersal patterns in San Joaquin kit fox and demographic factors, including population density, the number or sex ratio of adults, the sex ratios of juveniles, or the proportion of new juveniles in the population. They also did not find a relationship with ecological factors, including leporid density and total prey density, small mammal abundance, or coyote abundance.

Whether the spatial activity patterns exhibited by San Joaquin kit fox are applicable to desert kit fox is unknown, but it is likely that spatial activity by desert kit fox (i.e., territory and home range use, spacing, dispersal, etc.) is also dynamic and potentially related to demographic

and environmental factors such as prey availability (see discussion in Ecological Relationships).

Ecological Relationships

Fairly extensive research has been conducted on the ecological relationships of kit foxes to other species, and in particular to coyotes, which is a common predator of kit foxes (e.g., Rall and White 1995; White et al. 1995; White and Garrott 1997; Kozlowski et al. 2008) and direct competitor for prey (e.g., White et al. 1994, 1995; Arjo et al. 2007; Kozlowski et al. 2008). A brief summary of some of these studies, as they may relate to conservation of the desert kit fox in the Plan Area, is provided here.

Several studies have noted dramatic kit fox population fluctuations in relation to prey availability. For example, in San Joaquin kit fox, Cypher et al. (2000) found that high kangaroo rat densities positively influenced the growth of a kit fox population, while Moonjian (2007) found that low densities of kit foxes in the Palo Prieto area of western Kern County were associated with low densities of kangaroo rats. Local extirpations have also been linked to the previous loss of kangaroo rat populations (Cypher et al. 2000). White and Ralls (1993) found that prey scarcity related to drought reduced reproductive success in San Joaquin kit fox on the Carrizo Plain, with no reproduction by nine tracked females in 1990.

Prey selection by San Joaquin kit fox may also track availability. A 15-year study at the Naval Petroleum Reserves in western Kern County found that the dominant prey item alternated over time between kangaroo rats and leporids (Cypher et al. 2000). Similar prey studies have not been conducted for desert kit fox, but it is expected that patterns would be similar because desert rodent and lagomorph populations also vary substantially in relation to environmental conditions and possibly demographic factors. For example, Beatley (1969) found that desert rodent reproduction and population densities in southern Nevada were strongly associated with fall rain and production of winter annuals plants. Black-tailed jackrabbit densities and distribution appear to have a more complex relationship with environmental conditions because their diet shifts between seasons, locations, years, and vegetation types (Hayden 1966; Johnson and

Anderson 1984; Wansi et al. 1992). The length of the jackrabbit breeding season appears to be related to the production of herbaceous vegetation (Lechleitner 1959), and reproductive activity appears to be density-dependent, which can result in wide population fluctuations on 7–10-year cycles (French et al. 1965; Wagner and Stoddart 1972; Smith 1990).

Home-range size also appears to vary in relation to prey availability, with smaller home ranges where lagomorphs are abundant and larger home ranges when desert kit foxes have to rely on small prey such as kangaroo rats and other small rodents (Zoellick and Smith 1992).

Coyote are both predators of kit foxes and direct competitors for food, with substantial spatial, temporal, and dietary overlap (White et al. 1994, 1995; Kozlowski et al. 2008). Habitat and land use changes that attract coyotes therefore would likely have an adverse effect on desert kit foxes. Arjo et al. (2007), for example, suggest that invasion of a site in western Utah (the same site studied by Egoscue in the 1950s) by cheatgrass (*Bromus tectorum*), replacing native Great Basin shrub communities, and the addition of artificial water sources have altered prey abundance and attracted coyotes, to the detriment of kit foxes. Kit foxes do not require free water and are less water-limited than coyotes. The increased abundance of coyotes may have increased direct competition for food resources, with kit foxes having to focus on small rodents due to increased predation of lagomorphs by coyotes (Arjo et al. 2007). On the same Utah site, Kozlowski et al. (2008) found that kit foxes and coyotes used space within their home ranges differently, with kit foxes using areas of vegetation and ruggedness not favored by coyotes, but interactions were still common and 56% of kit fox mortalities were attributed to coyotes.

Population Status and Trends

Global: Apparently Secure (NatureServe 2012)
State: Not ranked
Within Plan Area: Not ranked

The desert kit fox currently does not have federal or California special status, although it is protected from hunting as a fur-bearing mammal under Title 14 of the California Code of Regulations, Section 460. Population status and trends in the Plan Area are unknown, but it has

been characterized as uncommon to rare in arid regions in California (Zeiner et al. 1990). Meany et al. (2006) state that kit fox populations "plummeted" in the last half of the 19th and early 20th century due to predator and rodent controls. They report that the kit fox population in Colorado may be close to extirpation, populations in Oregon and Idaho are extremely low, and populations in the Great Basin Desert in Nevada and Utah may be in decline. The only states Meaney et al. (2006) indicate may still have stable populations are Arizona, New Mexico, and Texas.

In March 2013 The Center for Biological Diversity (CBD) submitted a petition to the California Department of Fish and Wildlife (CDFW) to list the desert kit fox as threatened under the California Endangered Species Act (Kadaba et al. 2013). The CBD cited large-scale energy development as a primary threat, in concert with OHV use, grazing, agriculture, military activities, urbanization, climate change, and increased anthropogenic disease risks (Kadaba et al. 2013). Although the species' status and trends in the Plan Area are unknown, it is reasonable to assume that the threats and stressors cited in the CBD petition have resulted in loss, fragmentation, and degradation of habitat for kit fox in the Plan Area and at least local impacts on local populations subject to these threats and stressors (see Threats and Environmental Stressors). Whether these effects, as outlined in the petition, have risen to the level of warranting a listing as threatened is yet unknown and await analysis and determination by CDFW.

Threats and Environmental Stressors

An initial cause of population declines in kit fox was predator and rodent controls in the 19th and 20th centuries (Meaney et al. 2006). Several threat factors cited by Meaney et al. (2006) for Colorado that may apply to the desert kit fox in the Plan Area are habitat degradation, loss and fragmentation from development, roads, recreation, and grazing. The expansion and increased abundance of coyotes, which is the main predator of kit foxes, is also a threat.

A potentially devastating current threat to desert kit fox is canine distemper, which was determined to be the cause of death of several kit foxes at and near a solar energy project located west of Blythe in fall 2011 (Clifford et al. 2013). The source of the distemper outbreak

is not known and may have been a domestic dog or native wildlife such as badger. This distemper outbreak is the first documented incident in wild kit foxes (Clifford et al. 2013). Subsequent trapping of 39 individuals in January 2012 at the outbreak site found that all appeared healthy, but the capture rate at the affected site was low, indicating a reduction in the local population (Clifford et al. 2013). Although the recent outbreak of canine distemper is the first documented incident in wild kit foxes, O'Farrell and Gilbertson (1986) suggested that canine distemper or some other viral or bacterial disease may have been a causal factor in the apparent starvation deaths of several desert kit fox individuals during a study conducted from 1977 to 1979 in the western Mojave, because one clinical symptom of distemper is anorexia and gradual loss of activities, which can result in starvation. O'Farrell and Gilbertson (1986) observed that the animals died over a short time period, died underground, were emaciated and had no food in their gastrointestinal tracts, showed evidence of diarrhea, and had conjunctival secretions. Unfortunately the individuals were recovered too late for histopathological diagnosis (O'Farrell and Gilbertson 1986).

In addition to habitat impacts and disease, it is expected that desert kit fox is also vulnerable to various human activities, including recreation such as OHVs. However, O'Farrell and Gilbertson (1986) found that most dens were within 490–656 feet (150–200 meters) of roads or jeep trails in the Rand Open Area in the western Mojave that was subject to unlimited OHV activity during the study from 1977 to 1979 (i.e., there was no apparent tendency to locate dens away from roads or trails). However, mortalities related to shooting, vehicle collisions, den collapse (which could result from OHV activity), and potentially canine distemper (which could be transmitted by dogs) were observed.

In more urbanized areas, vehicle collisions are a frequent source of mortality of kit foxes. Bjurlin et al. (2005) found that vehicle collisions were the primary cause of mortality of San Joaquin kit foxes in the Bakersfield area, whereas predation is the more common cause of mortality of the subspecies in natural areas (e.g., Ralls and White 1995). Bjurlin et al. (2005) found that while kit foxes frequently crossed local roads, collisions were statistically more likely to occur on arterials with higher traffic densities and speeds; about 69% of all documented strikes were on four- and six-lane arterials and about

88% of all strikes were on roads with posted speed limits of 45, 50, or 55 miles per hour (56% of strikes were on roads with a 55-mile-per-hour speed limit). Bjurlin et al. (2005) also found that collisions on roads were disproportionate to males during the winter in association with territorial defense, mating, and exploratory movements. Further, even though den selection was not related to road proximity, close proximity of dens to roads increased collision risk.

Desert kit fox is also vulnerable to rodenticide poisoning (Shitoskey 1975; Meaney et al. 2006). Shitoskey (1975) demonstrated that three rodenticides—sodium monofluoroacetate (compound 1080), strychnine alkaloid, and zinc phosphate—were lethal to kit fox when administered directly. Sodium monofluoroacetate and strychnine alkaloid were also lethal when kit fox ingested kangaroo rats killed by the two rodenticides, but kit fox was able to tolerate kangaroo rats contaminated with zinc phosphate.

Military training will be an ongoing activity in the Plan Area, and noise associated with such activities, including from aircraft, may be a concern for overall stability of the desert kit fox, including potential direct effects on kit foxes and indirectly through effects on prey abundance and availability. Bowles et al. (1995) examined the effects of aircraft noise on kit fox and the desert rodent community on the Barry M. Goldwater Air Force Range in Arizona from 1991 to 1994. Monitoring on affected and control sites revealed no large differences in kit fox or rodent communities that could be attributed to aircraft noise, and observed differences between exposed and control population generally were within those expected through natural variability. Survival (as measured by "days known alive") for kit foxes on control and exposed sites were not significantly different, and the median survival days was actually higher on the exposed site at 223 days vs. 209 days for the control site. Individual weights (a measure of physical condition) and home-range sizes were also not different for the control and exposed sites. For the desert rodents, no statistical differences were found for species diversity, population densities, and weights (a measure of physical condition) between control and exposed sites. Annual rodent survival rates were higher in control sites, and recruitment was higher on exposed sites.

Conservation and Management Activities

The desert kit fox is not a special-status species, nor is it covered under any existing conservation plans in or adjacent to the Plan Area. It is not explicitly addressed in federal land use planning, such as the Bureau of Land Management's (BLM's) West Mojave Plan (2005), Northern and Eastern Colorado Plan (2002a), and Northern and Eastern Mojave Plan (2002b). It is also not explicitly addressed by the National Park Service general management plans for Mojave National Preserve, Death Valley National Park, and Joshua Tree National Park.

A management and monitoring plan for desert kit fox was developed for the Genesis Solar Energy Project (AECOM 2012) where several mortalities attributed to canine distemper occurred. The plan includes several avoidance and minimization measures for the project: pre-construction surveys; den classification and excavation of inactive den complexes in the construction area to prevent reuse; monitoring of potential and known active den complexes; exclusion of kit foxes from den complexes using passive methods; and protocols for handling sick, injured, or dead kit foxes.

Data Characterization

There is a lack of population and distributional information for desert kit fox in the Plan Area, including use of and movement through landscape. The local ecology of the species and the San Joaquin kit fox subspecies is well studied in some locales (e.g., western Utah, western Arizona, central California) with regard to life history traits and ecology, but only one older ecological study for the desert kit fox in the western Mojave portion of the Plan Area has been conducted (i.e., O'Farrell and Gilbertson 1986).

Management and Monitoring Considerations

Because suitable den sites may be a limiting resource for desert kit fox, maintaining suitable denning habitat may be important for conservation of the species, including relatively open habitat, gentle slopes, and friable soils (O'Farrell and Gilbertson 1986; Arjo et al. 2003). Other important factors may be conversion of habitats to annual grassland that could affect prey abundance and provision of

the artificial water sources that could attract coyotes that are predators of kit foxes and direct competitors for resources. The ability of kit foxes to move through the landscape may be enhanced by providing culverts in key locations.

Understanding causes of death is also an important management and monitoring consideration for desert kit fox, especially those with potential anthropogenic causes or interactions, including diseases such as canine distemper, vehicle collisions, and coyote predation and competition.

General ecological and behavioral studies for desert kit fox are also lacking for the Plan Area. Studies of other kit fox populations across the southwest reveal substantial variability in various life history traits, including habitat selection, demographics, predator–prey relationships, and vulnerability to various threats and stressors, suggesting that effective conservation and management of the desert kit fox in the Plan Area will require additional Plan-specific information.

In addition to maintaining suitable habitat and prey availability, mobility across the landscape is an important management and monitoring consideration, especially across roads that can be significant contributors to mortality. Kit foxes are known to cross highways at grade, but their use of below-grade crossings (e.g., culverts, bridges, and underpasses) is less understood. Boarman and Sazaki (1996) incidentally documented desert kit foxes activity at culverts under State Route (SR) 58 in the Plan Area approximately 7 miles east of Kramer Junction during a study of desert tortoise (*Gopherus agassizii*). The study observed kit fox activity around culverts, including steel pipes that were 2.9–4.9 feet (0.9–1.5 meters) in diameter, concrete pipes 55 inches (1.4 meters) in diameter, and concrete boxes 9.8–11.8 feet (3–3.6 meters) wide by 5.9–9.8 feet (1.8–3 meters) high, but it did not provide data documenting actual crossings using the culverts or whether culverts of certain dimensions were used or avoided. Due to telemetry equipment failures and low capture rates, a recent study of below-grade crossings of the four-lane SR-58 west of Barstow by desert kit foxes by Clevenger et al. (2010) was generally unsuccessful in documenting whether kit foxes cross the highway using available corrugated metal culverts, cement box culverts, and bridge crossings. Two individuals were documented successfully crossing the

highway, but it is unknown whether the crossings were through below-grade structures or at grade across the highway. However, Clevenger et al. (2010) did document two apparent swift fox crossings of Interstate 70 in Colorado using reinforced concrete pipe culverts and several crossings of Interstate 90 in South Dakota using culverts, including at least four two-lane and one four-lane crossing, and possibly a six-lane crossing. The dimension of the box culvert in the four-lane crossing was relatively tall and wide (84 x 84 inches) (Clevenger et al. 2010). A recent camera monitoring study for the Coachella Valley Multiple Species Habitat Conservation Plan (MSHCP) conducted at six highway underpasses in the Coachella Valley from September 2011 to April 2012 failed to detect any desert kit foxes (Murphy and Barrows 2012). However, the status and distribution of kit fox in the MSHCP plan area is unknown, so its apparent absence at the monitored underpasses is difficult to interpret.

Kit foxes in urbanized areas are known to cross roads, including six-lane arterials, but the risk of vehicle collisions is high on four- and six-lane arterials and was found to be the main cause of mortality in the Bakersfield area (Bjurlin et al. 2005). Bremner-Harrison et al. (2005) conducted a 1-year study of road culvert use in Kern County along Interstate 5, SR-14, and SR-58 and failed to document any use of culverts to cross roads. They hypothesized that kit foxes may associate the closed spaces of culverts with increased predation risk from coyotes, dogs, and bobcats (*Lynx rufus*) that were detected in and around crossing structures. They did not study use of large structures for crossing such as bridges over larger washes, and kit fox use of large structures is unknown. The diverse desert terrain in the Plan Area includes many culvert crossings under existing roads for drainage, but use of these culverts by desert kit fox is unknown (e.g., Clevenger et al. 2010), although O'Farrell and Gilbertson (1986) documented use of a road culvert as a den.

Bremner-Harrison et al. (2005) made several recommendations regarding road crossings for kit fox that may be applicable to the Plan Area:

1. Conduct further field investigations to determine whether kit foxes are indeed avoiding structures and crossing roads, or are generally avoiding roads.

2. If opportunities arise, repeat this investigation in areas with median barriers to determine whether kit foxes are more likely to use crossing structures in such areas or simply abandon attempts to cross roads.
3. In areas where median barriers are present along highways, recommendations to reduce adverse impacts to kit foxes include:
    a. install fencing to exclude kit foxes from the highway and direct them to crossing structures;
    b. design crossing structures to accommodate use by the largest animal species occurring in the local ecosystem, and
    c. place artificial dens within crossing structures and near entrances to provide escape cover for kit foxes. (Bremner-Harrison et al. 2005, p. 42)

Based on other known and likely threats and stressors to kit fox in the Plan Area, other management and monitoring considerations include:

- Developing demographic data for desert kit fox in the Plan Area, including population size and distribution
- Understanding the ecological relationships between kit foxes and coyotes in the Plan Area
- Understanding predator/prey relationships and maintaining and enhancing prey populations in areas supporting kit foxes
- Managing the use of rodenticides and other pesticides
- Managing and monitoring the incidence of diseases such as canine distemper.

Predicted Species Distribution in Plan Area

This section provides the results of habitat modeling for desert kit fox, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 15,686,640 acres of modeled suitable habitat for desert kit fox in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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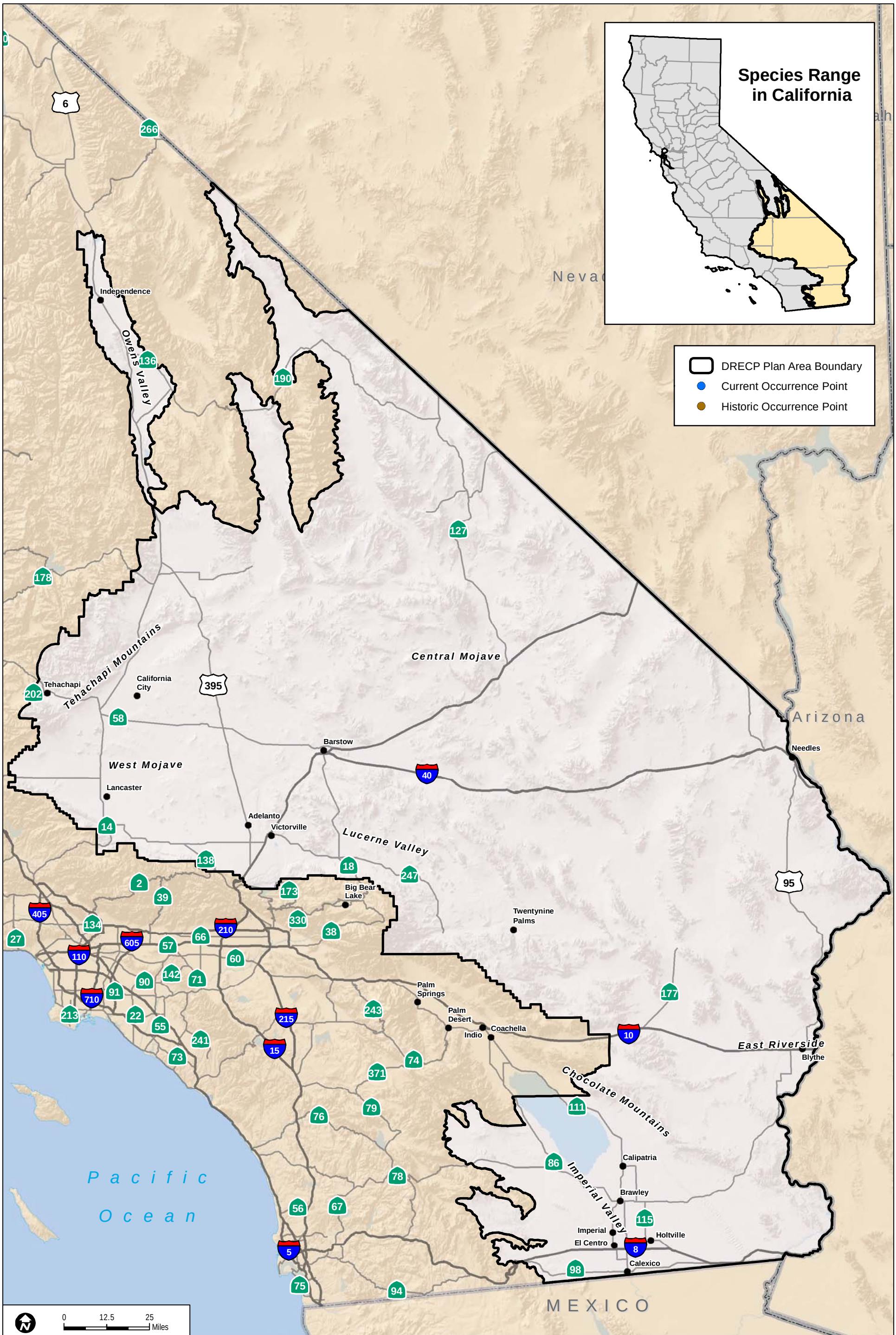

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M04

Desert Kit Fox Occurrences in the Plan Area

Mohave Ground Squirrel (*Xerospermophilus mohavensis*)

Legal Status

Photo courtesy of Phil Leitner.

State: Threatened
Federal: None
Critical Habitat: N/A
Recovery Planning: No formal state or federal recovery plans have been prepared.
Note: The U.S. Fish and Wildlife Service (USFWS) published a 12-month finding on October 6, 2011, that listing of the Mohave ground squirrel (*Xerospermophilus mohavensis*) is not warranted at this time (76 FR 62214–62258).

Taxonomy

The Mohave ground squirrel (*Xerospermophilus mohavensis*) was discovered by F. Stephens in 1886 and described as a distinct monotypic species by Merriam in 1889. The type locality is the Mohave Desert near Rabbit Springs, about 24 kilometers (15 miles) east of Hesperia in San Bernardino County (Helgen et al. 2009).

The Mohave ground squirrel belongs to the family Sciuridae, which includes rodents that dig their own burrows (Gustafson 1993). Previously recognized as *Spermophilus mohavensis*, based on a review of morphometrics (measurement of external form and structure) and molecular phylogenetics (evolutionary relationships within and between groups), the Mohave ground squirrel is now recognized as *Xerospermophilus mohavensis* (Helgen et al. 2009). The Mohave ground squirrel is a distinct, full species with no recognized subspecies (Helgen et al. 2009). However, there has been some question about the recognition of the round-tailed ground squirrel (*Xerospermophilus tereticaudus*) and the Mohave ground squirrel as distinct species (Gustafson 1993; Hafner 1992; Hafner and Yates 1983). The two squirrels are closely related and have a contiguous,

but not overlapping, geographic range (Best 1995; Hafner 1992). Hafner and Yates (1983) described a narrow hybridization zone in the ranges of the two species in an area northwest of Helendale and near Coyote Dry Lake northeast of Barstow, but studies by Hafner and Yates (1983) and Hafner (1992) demonstrated that there were sufficient chromosomal, genetic, morphological, and ecological differences to warrant distinct species recognition.

Distribution

General

Endemic to California, the Mohave ground squirrel is exclusively found in the northwestern Mojave Desert in San Bernardino, Los Angeles, Kern, and Inyo counties (Best 1995; Figure SP-M05).

Distribution and Occurrences within the Plan Area

Historical

The presumed historical range of the Mohave ground squirrel within the northwestern Mojave Desert was bounded on the south and west by the San Gabriel, Tehachapi, and Sierra Nevada mountain ranges; on the northwest by Owens Lake, and on the northeast by the Granite and Avawatz mountains; and on the east and southeast by the Mojave River (Leitner 2008; MGSWG 2011). In addition, the species was historically found in one locality east of the Mojave River in the Lucerne Valley. Its historic range covered about 20,000 square kilometers (km^2) (7,722 square miles [mi^2]) (Gustafson 1993), which is the smallest geographic range of any ground squirrel species in the United States. However, for the 12-month finding for the species published in October 2011, USFWS used a somewhat larger historical range of approximately 21,525 km^2 (8,311 mi^2) (76 FR 62214–62258). USFWS also stated in the 12-month finding that the range of the Mohave ground squirrel may be larger than defined in the finding or previously published based on recent sightings such as in an interior valley of the Tehachapi Mountains and in the Panamint Valley about 8 kilometers (5 miles) north of the defined range (76 FR 62214–62258).

Based on the range used by Leitner (2008), about 88% of the historical range of the species is within the Plan Area (only the Coso Range in the northern extent of its historic range is excluded).

Prior to conversion of native desert habitats in the Antelope Valley west of Palmdale and Lancaster to agriculture and residential and commercial development, there was potential habitat for the Mohave ground squirrel, but there are no historical or recent occurrence records in this area west of State Route 14 (Leitner, pers. comm. 2012).

Approximately 28% of the California Natural Diversity Database (CNDDB) records for the Mohave ground squirrel are historical or have no date. These records are located throughout the species' range (Figure SP-M05) (CDFW 2013).

Recent

The current range may be reduced from the historical range as a result of the possible extirpation of the Mohave ground squirrel in the western portion of the Antelope Valley; although there is suitable desert scrub, there are no historical records for areas west of State Route 14. The species has been extirpated from much of the Victorville area due to agricultural and more recent rapid urban development, but there are a few recent CNDDB records, including from 2005, 2007, and 2011, for the Adelanto area (CDFW 2013; Dudek 2013; Figure SP-M05), indicating a possible relict population in the southern portion of its range (Leitner, pers. comm. 2012).

Habitat for the species has been reduced by development of agricultural uses, grazing, urbanization, military activities, energy production, and recreation (MGSWG 2011). The current occupied range is estimated to be about 19,000 km^2 (6,640 mi^2) (MGSWG 2011).

The occurrence of Mohave ground squirrel is likely to be patchy within its range, even within apparently suitable habitat (MGSWG 2011). However, as noted by Leitner (2008), occurrence records tend to be concentrated in certain areas where trapping studies have been focused; these studies are discussed in more detail below. There has not been a systematic, range-wide census or statistically based random sampling study to determine occupation throughout the

species' range (Leitner 2008). About 88% of the geographic area of known existing populations of the species, based on Leitner (2008), occur in the Plan Area (only a portion of the Coso Range-Olancha Core population is outside this area).

Recent (after 1990) records from the CNDDB and West Mojave Plan Mohave ground squirrel transect data and other California Department of Fish and Wildlife (CDFW) data include location occurrences ranging from Inyo County in the north to 3 miles southwest of Rabbit Lake in the south. The eastern extent ranges to the Granite Mountains and Fort Irwin and the westernmost record is just east of Oak Creek (Figure SP-M05) (Dudek 2013).

Leitner (2008) provides the most current status of the Mohave ground squirrel based on compilation of a database, including unpublished field studies, surveys, and incidental observations for the 10-year period from 1998 through 2007 (Table 1). This database includes 1,140 trapping sessions, of which 102 resulted in observation of the species, and 96 additional incidental observations. Most of these studies and observations have been conducted in the southern part of the species' range south of State Route 58 and no range-wide systematic or statistically based random sampling has been conducted to characterize the species' status throughout its range. Leitner (2008) emphasizes that there are large areas of potential habitat where the species' status is unknown, especially on the China Lake Naval Air Weapons Station and Fort Irwin.

Table 1. Mohave Ground Squirrel Regional Occurrence Information

Regional Location	Data Summary
Inyo County between Olancha and Haiwee Reservoir, Coso Range within China Lake Naval Air Weapons Station	Detected on five trapping grids, including Lee Flat just inside Death Valley and the northernmost occurrence record. Four other incidental records, including in north Panamint Valley several kilometers north of generally accepted range.
Ridgecrest area	Detected on 5 of 10 trapping grids in vicinity of Ridgecrest and 6 of 10 grids along State Route 176 east of Ridgecrest. No individuals trapped at two sites in Spangler Hills southeast of

Table 1. Mohave Ground Squirrel Regional Occurrence Information

Regional Location	Data Summary
	Ridgecrest.
Little Dixie Wash extending from Inyokern southwest to Red Rock Canyon State Park	Detected on 6 of 7 trapping grids scattered throughout valley and more than 20 incidental observations. Species widespread in area.
Fremont Valley to Edwards Air Force Base	No detections in last 10 years on 6 trapping grids in Fremont Valley. Thirteen records around periphery of Desert Tortoise Natural Area (DTNA) and likely to be present within DTNA. Two incidental records northeast of town of Mojave, but protocol trapping studies in area have been negative. Ten trapping and incidental observation records for area north of Boron and Kramer Junction. Species likely widespread across region.
Wind farm southwest of Mojave (outside accepted range but appears to have suitable habitat)	No detections at 24 trapping grids southwest of town of Mojave. Two unconfirmed observations in CNDDB.
Edwards Air Force Base	Extensive monitoring conducted, with 6 observations on 40 trapping grids from 2003–2007. Distribution of species on Edwards Air Force Base is well documented.
Los Angeles County desert area	No detections on 52 trapping grids. Four positive records in small area near Rogers Dry Lake on Edwards Air Force Base.
Victor Valley to Barstow	Extensive surveys of Adelanto and western Victorville area with two trapping records and one incidental observation. One capture near intersection of U.S. 395 and I-15. These records indicate small residual population in area. No records east of Mojave River since 1955, but not well sampled in last 10 years. No detections on three trapping sites from El Mirage Dry Lake north and east toward Barstow.

Table 1. Mohave Ground Squirrel Regional Occurrence Information

Regional Location	Data Summary
Barstow area	Three records – one record about 3.5 miles south of Barstow near landfill and outside accepted range and two records west of City. One detected at the edge of alfalfa field near Harper Dry Lake and the other trapped about 6.1 miles west of Hinkley near State Route 58.
Coolgardie Mesa and Superior Valley north of Barstow	Positive records for three trapping grids and at least seven incidental observations.
Pilot Knob area	Detected five sites from Cuddeback Dry Lake east to the boundary of the China Lake Naval Air Weapons Station.

Source: Leitner 2008.

Approximately 52% of the CNDDB records are located on public lands managed by the BLM, Department of Defense, California Department of Transportation, Department of Parks and Recreation, Kern and San Bernardino counties, and the Los Angeles Department of Water and Power). Approximately 21% are located on privately owned lands. The ownership of the remaining 27% of the CNDDB records is unknown (CDFW 2013).

Natural History

Habitat Requirements

The Mohave ground squirrel occurs in a variety of desert shrubland habitats (Table 2). Although most often found in creosote bush scrub, it has also been recorded in desert saltbush scrub, desert sink scrub, desert greasewood scrub, shadscale scrub, Joshua tree woodland, and Mojave mixed woody scrub (Best 1995; 75 FR 22063–22070; MGSWG 2011). Mohave ground squirrel typically occupies areas with open vegetative cover and small bushes (< 0.6 meter (2 feet) in height) spaced approximately 6 to 9 meters (20 to 30 feet) apart (Best 1995).

Table 2. Habitat Associations for Mohave Ground Squirrel

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Creosote bush scrub, Desert saltbush scrub, Desert sink scrub, Desert greasewood scrub, Shadscale scrub, Joshua Tree woodland, Mojave mixed woody scrub	Primary habitat	Active and Inactive Season	Deep, sandy to gravelly soils on flat to moderately sloping terrain with open vegetative cover	Best 1995; MGSWG 2011

Mohave ground squirrel prefers deep, sandy to gravelly soils on flat to moderately sloping terrain and will avoid rocky areas for the most part (Best 1995; MGSWG 2011). The species is not known to occupy areas of desert pavement (MGSWG 2011). Soil characteristics are particularly important because Mohave ground squirrels construct burrows to provide temperature regulation, avoid predators, and use during the inactive season (75 FR 22063–22070).

Foraging Requirements

The Mohave ground squirrel primarily feeds on plant material. In the short term, they specialize in foraging on certain plant species, but as these sources become less available throughout the active season, the Mohave ground squirrel adapts its foraging strategy to maximize energy intake, exploiting food sources that are intermittently available (75 FR 22063–22070). High water content may be a component of their food selection as plants are eaten at different times depending on their water content (Best 1995; 75 FR 22063–22070). Mohave ground squirrels consume the leaves, fruits, and seeds of a variety of annual and perennial plants, fungi, arthropods, including butterfly larvae. At various times of the year and depending on location, they may consume leaves, forbs, shrubs, and grasses of several species and genera, including creosote (*Larrea tridentata*), winter fat (*Krascheninnikovia lanata*), spiny hop-sage (*Grayia spinosa*), freckled milk-vetch (*Astragalus lentiginosus*), eremalche (*Eremalche exilis*), desert-marigold (*Baileya pleniradiata*),

langloisia (*Langloisia setosissima*), Mojave monardella (*Monardella exilis*), saltbush (*Atriplex* spp.), gilia (*Gilia* spp.), golden linanthus (*Linanthus aureus*), and Mediterranean grass (*Schismus arabicus*), as well as seeds of box thorn (*Lycium* spp.) (Best 1995; 75 FR 22063–22070; MGSWG 2011). On the Coso Range (outside of the Plan Area), about 42% of the species' diet, based on fecal samples, consisted of forbs and shrub material (primarily foliage) (MGSWG 2011). Shrubs are especially important both early and late in the active season when forbs are not available (MGSWG 2011). Winter fat, spiny hop-sage, and saltbush made up 60% of the species' shrub diet, indicating that these species are the main food source when forbs are unavailable (MGSWG 2011). It has been suggested that habitats where winter fat and hop-sage are absent may be suboptimal for Mohave ground squirrel (MGSWG 2011).

Reproduction

The Mohave ground squirrel breeding season is from mid-February to mid-March (Best 1995; Laabs 2006) (Table 3). Males emerge from hibernation in February, up to two weeks before females, and during this time they may be territorial (Best 1995). Females generally only occupy male territories for one or two days then establish their own home ranges after copulation. Recent radiotelemetry data indicate that males expand their activity areas the breeding to overlap several established female ranges, (unpublished data, Leitner, pers. comm. 2012). Males stake out the overwintering sites of females to mate with them when they emerge (MGSWG 2011).

Pregnant females are present from March through April (Leitner, pers. comm. 2012) and gestation lasts from 29 to 30 days (Best 1995). Litter sizes range from four to nine (Best 1995), though mortality of juveniles is high during the first year, especially for juvenile males (MGSWG 2011). Parental care and lactation continues through mid-May. Litters generally appear above ground in early May (Harris and Leitner 2004). Females will breed at 1 year of age if environmental conditions are suitable, but males do not mate until 2 years of age (MGSWG 2011).

The amount of fall and winter precipitation generally determines Mohave ground squirrel reproductive success. In low rainfall years (e.g., less than 6.5 cm [2.6 in.]), they may forego breeding (MGSWG 2011), and breeding may not occur for several years during prolonged

drought (Best 1995). Because of the small geographic range of the species, low rainfall can lead to reproductive failure throughout the range (MGSWG 2011). During these periods, all available forage may be converted to body fat and squirrels can enter dormancy as early as April (Leitner 1999).

Table 3. Key Seasonal Periods for Mohave Ground Squirrel

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Aestivation								X	X	X		
Hibernation	X										X	X
Breeding		X	X									
Parental Care			X	X	X							

Notes: Aestivation is the summer period of inactivity and hibernation is the winter period of inactivity.

Sources: Best 1995; Laabs 2006.

Spatial Behavior

The Mohave ground squirrel is generally only active above ground between February and July (MGSWG 2011), but the active period may begin as early as mid-January (Harris and Leitner 2004). Adults generally enter aestivation earlier than juveniles (MGSWG 2011). Timing of emergence varies geographically as it appears to depend on temperature and elevation (Gustafson 1993; Laabs 2006). Furthermore, the timing of emergence and length of the active season varies by sex, age, and availability of food resources (MGSWG 2011). Adult females and juveniles generally have longer active seasons than adult males. The active season is also longer when there is more food available, which is often correlated with greater precipitation (MGSWG 2011). Mohave ground squirrels are diurnal, spending much of the day above ground during the active season. During the inactive season, Mohave ground squirrels remain underground in burrows and enter a state of torpor (a state of reduced physiological activity or sluggishness) to conserve their energy reserves and water (Best 1995; MGSWG 2011).

Harris and Leitner (2004) conducted a 5-year radiotelemetry study of home range use by Mohave ground squirrels in the Coso Range in Inyo County. At this study site, individual Mohave ground squirrel home ranges (calculated using both minimum convex polygon and adaptive kernel methods) varied substantially by year, individual, sex, and season (i.e., mating season vs. post-mating season) (Table 4). Generally, males have larger home ranges than females, with the most pronounced differences during the mating season. Female ranges expanded during the postmating season compared to the mating season (Table 4). In drought years when reproduction did not occur, female postmating season home ranges varied inversely in relation to precipitation, which in turn is related to the amount of available forage (Harris and Leitner 2004). Female home ranges contracted in years of moderate drought and lack of reproduction, which may be a strategy to reduce energy expenditure and enter dormancy sooner (Harris and Leitner 2004). During years of high precipitation and successful reproduction, female postmating home ranges were larger in response to the need for more energy sources to support gestation and lactation (Harris and Leitner 2004). Females that were radio tracked for more than 1 year showed a high level of home range site fidelity and all individuals' home ranges exhibited overlap over different years; i.e., no females moved to entirely new home ranges (Harris and Leitner 2004).

Table 4. Mohave Ground Squirrel Home Ranges in the Coso Range[1]

Type	Median MCP Home Range[2]	Citation
Mating Season Home Range – Male	16.63 acres (range: 10.5–99.1 acres)	Harris and Leitner 2004
Mating Season Home Range - Female	1.83 acres (range: 0.70–2.3 acres)	Harris and Leitner 2004
Postmating Home Range – Male	3.06 acres[3]	FR 22063–22070
Postmating Home Range – Female	2.96 acres[3]	FR 22063–22070

Notes:
[1] The Coso Range is located north of the Plan Area
[2] MCP = minimum convex polygon
[3] The home range statistics reported in FR 22063–22070 (the 90-day finding on the petition to list the species) cite Harris and Leitner (2004), but the original paper does not appear to include these specific statistics for postmating home ranges. While these statistics appear to be consistent with Figure 1 in Harris and Leitner (2004) and are consistent with the text description of postmating home ranges, they cannot be confirmed by a review of the original paper and it is unclear how these statistics were generated for the 90-day finding on the petition.

Male home ranges during the mating season were very large and reflected long-distance movements large enough to cross the home ranges of several females (Harris and Leitner 2004). Long-distance movements (> 656 feet) were much more frequent during the mating season compared to the postmating season, and females seldom made such long movements (Harris and Leitner 2004).

Mohave ground squirrels maintain three types of burrows within their home ranges: (1) home burrows that are used overnight during the active season and usually located at the edge of a home range; (2) aestivation burrows; and (3) accessory burrows that are used during social interactions or for escape and thermoregulation during the midday (Best 1995). Burrows are typically constructed under large shrubs (MGSWG 2011).

Harris and Leitner (2005) used radiotelemetry to track dispersal movements by juvenile Mohave ground squirrels in their first year to hibernation sites. Most juveniles dispersed relatively long distances from their natal burrow area, and exhibited dispersal that is farther than other squirrels and other mammals in proportion to home range sizes (Harris and Leitner 2005). Mean male dispersal from the natal area was 9,580 feet (range: 0 to 20,439 feet) and mean female dispersal from the natal area was 2,470 feet (range: 0 to 12,670 feet) (Harris and Leitner 2005). However, with the exception of the one female that moved 12,760 feet to a hibernation site, all the females dispersed less than 1,640 feet from the natal area, indicating that juvenile dispersal is male-biased (Harris and Leitner 2005). Notably, the juveniles that dispersed more than 2,160 feet moved out of the alluvial basin where the study was located and had to cross rocky terrain with low shrub cover, which is not considered suitable habitat for the species, and at least two individuals crossed dirt roads (Harris and Leitner 2005). In addition, all but one of the individuals dispersing more than 2,160 feet left the natal area on a particular day and did not return to the natal area (Harris and Leitner 2005). Shorter dispersal movements may involve exploratory movements where juveniles return to the natal area at night before a permanent move. Harris and Leitner (2005) suggest that the relatively mobile behavior of juvenile Mohave ground squirrels may have adaptive value for connecting location populations and recolonizing sites that have experienced natural local extinctions (e.g., due to prolonged drought).

Ecological Relationships

There is little direct information on the potential role of Mohave ground squirrels in maintaining ecological relationships and processes. Their burrow systems likely provide refuge for other species that do not dig their own burrows such as snakes and lizards and potentially other small rodents. The range of the Mohave ground squirrel is entirely overlapped by the diurnal white-tailed antelope squirrel (*Ammospermophilus leucurus*), but there appears to be little direct competition between the two species (MGSWG 2011). While Mohave ground squirrels primarily forage on the foliage of shrubs and forbs, and secondarily on the seeds of shrubs and forbs, the antelope squirrel exhibits the opposite behavior of concentrating on seeds of forbs and shrubs and insects (about 25% of their diet) and secondarily foraging on foliage (MGSWG 2011). The Mohave ground squirrel is behaviorally dominant over the antelope squirrel (MGSWG 2011). As primarily a seed-eater, the antelope squirrel is also active on the surface year round (MGSWG 2011). Potential competitive relationships with birds, herbivorous reptiles (e.g., desert tortoise), or ants for food resources are unknown. They are probably prey for several natural predators, such as coyote (*Canis latrans*), American badger (*Taxidea taxus*), bobcat (*Lynx rufus*), red-tailed hawk (*Buteo jamaicensis*), golden eagle (*Aquila chrysaetos*), prairie falcon (*Falco mexicanus*), common raven (*Corvus corax*), and Mojave rattlesnake (*Crotalus scutulatus*) (Best 1995).

Population Status and Trends

Global: Moderate decline to relatively stable (NatureServe 2011)
State: Same as above
Within Plan Area: Same as above

Data are lacking to assess population abundance and trends for the Mohave ground squirrel (76 FR 62219). Systematic or sample-based surveys in the species' range have not been conducted at a level that allow for population estimates and comparisons over time. As discussed in Distribution, the species likely has been extirpated from portions of its former range due to urban and agricultural development, especially around the Lancaster, Palmdale, and Victorville areas.

Threats and Environmental Stressors

The primary threat to the Mohave ground squirrel has been habitat loss and fragmentation (Leitner 2008; MGSWG 2011). The Mohave ground squirrel's range has been reduced or its habitat destroyed and degraded by urban and rural development on private and public lands, agricultural development, military activities, energy projects, and transportation (Leitner 2008; MGSWG 2011; 76 FR 62214–62258). For energy projects, large-scale solar projects are particularly destructive to Mohave ground squirrel habitat because they have a large disturbance footprint and they are sited on level and gently sloping terrain that is characteristic of Mohave ground squirrel habitat (76 FR 62214–62258).

Livestock grazing and off-highway vehicles (OHVs) may also cause habitat degradation and have direct impacts on Mohave ground squirrel (Leitner 2008; MGSWG 2011; 76 FR 62214–62258).

Grazing by cattle and sheep can affect vegetative structure, disturb soils, accelerate erosion, and collapse burrows (MGSWG 2011). Cattle and sheep forage on winter fat foliage, which is also important to Mohave ground squirrel, especially in years with low precipitation and annual forb production (MGSWG 2011). Although livestock grazing is listed as a potential threat to Mohave ground squirrel, the BLM has been eliminating or reducing grazing in some areas of the species range (76 FR 62237) and grazing does not occur on military lands, state parks or CDFW ecological reserves (Leitner, pers. comm. 2012). The USFWS 12-month finding on October 6, 2011 conclude that livestock grazing is not currently a threat to the Mohave ground squirrel (76 FR 62214–62258).

OHV use is a threat to Mohave ground squirrel through direct collisions, disturbance of soil, destruction of shrubs, and facilitation of invasive species that displace native species along dirt roads and trails (MGSWG 2011). The West Mojave Plan Route Designation report indicates that 47% of 310 vegetation transects are bisected by some type of off-road vehicle track (MGSWG 2011). The four BLM-operated off-highway areas (Jawbone Canyon, Dove Springs, El Mirage, and Spangler Hills) cover over 417 km^2 (161 mi^2) within the Mohave ground squirrel's range (MGSWG 2011).

Prolonged drought is another threat to the Mohave ground squirrel. Low rainfall causes reduced productivity of annual plants, which can cause Mohave ground squirrels to forego breeding during drought periods because insufficient energy is available to support gestation and lactation (Best 1995; Harris and Leitner 2004). Local population extinction can result with prolonged drought events that suppress reproduction for several years (Best 1995). Prolonged drought events alone would not pose a serious threat to the species, considering its likely adaptations for these conditions, such as prolonged aestivation and long dispersal movements that allow for recolonization (Best 1995; Harris and Leitner 2005). However, habitat loss, fragmentation, and degradation can preclude recolonization of habitat from which local populations have been extirpated as a result of drought because the sites become functionally isolated from occupied areas (Laabs 2006).

Urban and rural uses have introduced potential impacts to Mohave ground squirrel that may occur where habitat is near development. Domestic cats (*Felis catus*) and dogs (*Canis familiaris*) may be predators and the use of rodenticides and pesticides around agricultural fields, golf courses, earthen dams, and canal levees may directly affect the species (MGSWG 2011).

Although common raven is a natural predator, their populations have increased substantially within the Mohave ground squirrel's range and they are a known predator for small mammals (MGSWG 2011). Therefore, ravens may be exerting higher predation pressure on the species than occurred historically.

Conservation and Management Activities

Conservation and management planning for the Mohave ground squirrel has been ongoing on several fronts, including by the West Mojave Plan; CDFW; the Desert Managers MGSWG; and on military installations.

The West Mojave Plan establishes a 1,726,712-acre (2,698 mi^2) Mohave ground squirrel Conservation Area on non-military public and private lands for the long-term survival and protection of the species. The Conservation Area covers about 41% of the estimated current range of the species. Public lands within the Conservation Area would be designated as a BLM Wildlife Habitat Management

Area. The West Mojave Plan established two goals for Mohave ground squirrel: Goal 1, ensure long-term protection of Mohave ground squirrel habitat throughout the species' range; and Goal 2, ensure long-term viability of the species throughout its range. The West Mojave Plan also established several objectives to meet these goals.

For Goal 1, the West Mojave Plan objectives are:

- Establish a Conservation Area for the protection of unfragmented habitat outside military installations (noted previously)
- Establish biological transition areas to minimize indirect impacts of human development on the Conservation Area
- Allow for adjustment of the Conservation Area boundary based on scientific studies
- Implement actions to ensure long-term protection of habitat for Mohave ground squirrel in the Conservation Area throughout the life of the Plan
- Annually track the loss of Mohave ground squirrel habitat resulting from Plan implementation
- Cooperate with military installations in sharing scientific information and reviewing management plans to assist managers in evaluating Mohave ground squirrel habitat protection on the installations.

For Goal 2, the West Mojave Plan objectives are:

- Per CDFW mandate, minimize and fully mitigate the impacts of the Plan's incidental take of Mohave ground squirrel throughout the life of the Plan
- Upon Plan adoption, implement studies that would determine four measureable biological parameters for the Mohave ground squirrel: (1) regional status; (2) potential "hot spots" (refugia); (3) genetic variation throughout the species' range; and (4) the species' ecological requirements
- Establish long-term study plots throughout the species' range to annually monitor populations, and fund continued

monitoring in the Coso Range (outside of the Plan Area) to provide baseline information

- Use the biological information from the above objectives to modify management prescriptions, as warranted, to ensure the long-term viability of the species.

To date, CDFW has spent approximately $800,000 funding studies that include information on genetics, diet, dispersal, and location of Mohave ground squirrels over the past several years. Also, approximately $100,000 from Section 2081 incidental permits has or will fund Mohave ground squirrel trapping administered by the Desert Tortoise Preserve Committee (MGSWG 2011).

The military has also conducted activities to inform conservation and management of the Mohave ground squirrel.

Edwards Air Force Base has completed at least 3 years of Mohave ground squirrel inventories and has monitored 60 Habitat Quality Analysis plots. Since 2003, approximately 45% of the Edwards Air Force Base has been surveyed and funds are programmed for Mohave ground squirrel inventories through 2013 (MGSWG 2011).

The National Training Center (NTC) and Fort Irwin contain 445,241 acres of Mohave ground squirrel habitat. The NTC and Fort Irwin funded trapping studies for the Mohave ground squirrel in 1977, 1985, and from 1993 to 1994. The MGSWS (2011) suggests that the three conservation areas for Lane Mountain milk-vetch (*Astragalus jaegerianus*) on Fort Irwin will work well for Mohave ground squirrel conservation. In addition, under an agreement with CDFW, the Paradise Conservation Area will be enhanced for Mohave ground squirrel by planting the species' preferred food plants (MGSWG 2011). However, at present there is no evidence that these areas support the Mohave ground and, further, these areas are generally rocky and hilly with little of the alluvial soils needed by the species (Leitner, pers. comm. 2012). There is currently no evidence that food enhancement is successful in the Paradise Conservation Area (Leitner, pers. comm. 2012).

Data Characterization

Because Mohave ground squirrel is inactive much of the year, and squirrel abundance and the length of the active season varies from year to year (MGSWG 2011), even when studies are scheduled carefully they may not be able to establish the presence or absence of the species from a site with a high level of certainty. Further, if unfavorable conditions (little fall and winter precipitation) persist for several seasons, local extirpation can occur, but re-colonization of these areas under more favorable conditions can occur. In addition, the species is not distributed continuously throughout its range independent of proposed habitat conversion (MGSWG 2011). Because trapping studies typically are sited in habitat proposed for conversion, grids and transects are not randomly or systematically placed in a manner that samples across the range of potentially suitable habitats and allows for inferences about occupation throughout the species' range. Many of the trapping studies for Mohave ground squirrel have been concentrated south of State Route 58 where most of the habitat conversion has been proposed (Leitner 2008). For this reason, there are extensive areas of the Mohave ground squirrel's range in the Plan Area that have not been studied and the species' status is unknown (Leitner 2008).

Management and Monitoring Considerations

Protection of large core areas of native habitat and adequate connections among the core areas are required to ensure the long-term survival and recovery of the Mohave ground squirrel. Ideally, biological, demographic, and genetic considerations should govern the size and location of preserve areas. As an initial recommendation for habitat conservation of currently occupied habitat, Leitner (2008) defines core areas for the species based on three objective and measureable criteria:

1. Demonstrated species persistence in an area over a long time period on the order of two to three decades;
2. Species must be currently present in multiple locations within the core area; and
3. There are substantial numbers of adults forming a viable reproductive population.

With these criteria in mind, core preserve areas need to be large enough to support populations that are resilient to natural fluctuations in size that occur in relation to precipitation patterns, including prolonged drought. Each population has to be large enough to withstand several years of no or reduced reproduction; if a drought extends so long that no reproduction occurs over a 4- or 5-year period, even the youngest cohort would likely die of old age before reproducing. Therefore, large preserve areas are needed to minimize the risk of local extinction from demographic and environmental stochastic events, as well as from the genetic problems associated with small population size, such as loss of genetic variability, genetic drift, and inbreeding depression. Smaller areas are also more susceptible to edge effects and disturbance from surrounding non-compatible land use (Laabs 2006).

Core reserves in high-quality habitats are required to support populations of the species during drought conditions and that can provide sources from which populations may expand when conditions are favorable to the species. Research conducted on the Coso Range (outside of the Plan Area) found that certain shrub species (winter fat and spiny hop-sage) appear to be important in providing forage when annual forb growth is low and thus may be critical to the persistence of populations during drought years (MGSWG 2011). However, these data are primarily from a study site at the north edge of the species' range and community (Mojave Mixed Woody Scrub) that is somewhat atypical of the majority of the species' range. Additional research into food habits and critical habitat features in creosote bush scrub and saltbush scrub habitats is needed to identify critical habitat features (Laabs 2006).

Based on the three objective criteria cited previously, Leitner (2008) identified four core areas, as summarized in Table 5. It is important to note that these core areas are only those identified so far and that with more survey data other areas may meet the objective criteria for a core area (Leitner 2008).

Table 5. Mohave Ground Squirrel Core Areas

Core Area Name	Area (acres)	Number of Positive Records (1998–2007)
Coso/Olancha	111,690	33
Little Dixie Wash	97,112	44
Coolgardie Mesa/Superior Valley	127,450	23
Edwards Air Force Base	76,761	34

Source: Leitner 2008.

As a rare species with apparent disjunct local populations, preserving naturally occurring genetic variability is critical to the preservation of the Mohave ground squirrel. Connectivity between preserve areas will be important to maintain gene flow between local populations and facilitate recolonization of areas if local extinctions occur. According to Leitner (2008), the four core areas identified are isolated from each other by distances that range from 30 to 50 miles. Leitner (2008) identified conceptual linkages between the corridors. Demographic considerations, such as home range size and average dispersal distances, should determine the width of connectivity corridors (Laabs 2006). As described previously, Mohave ground squirrels are capable of dispersing relatively long distances; the maximum juvenile male dispersal was about 3.9 miles and the maximum female dispersal was about 2.4 miles (Harris and Leitner 2005). With distances between core habitat areas of 30 to 50 miles (Leitner 2008), substantial swaths of suitable habitat between core areas will therefore be necessary.

The habitat management component of the *Draft Mohave Ground Squirrel Conservation Strategy* (MGSWG 2011) focuses on limiting habitat loss through effective conservation measures, mitigation, and compensation by avoiding and minimizing impacts to Mohave ground squirrel and its habitat and restoring and enhancing habitat. The strategy also focuses on securing and managing sufficient core habitat and corridors to maintain self-sustaining populations (MGSWG 2011). The West Mojave Plan also focuses on establishing conservation areas to protect unfragmented habitat and biological transition areas to

protect conservation areas from indirect human impacts. The West Mojave Plan includes objectives for implementing biological studies regarding the species' range, hot spots, and ecological requirements. This information would be used to inform conservation and management of the species.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Mohave ground squirrel, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 3,501,554 acres of modeled suitable habitat for Mohave ground squirrel in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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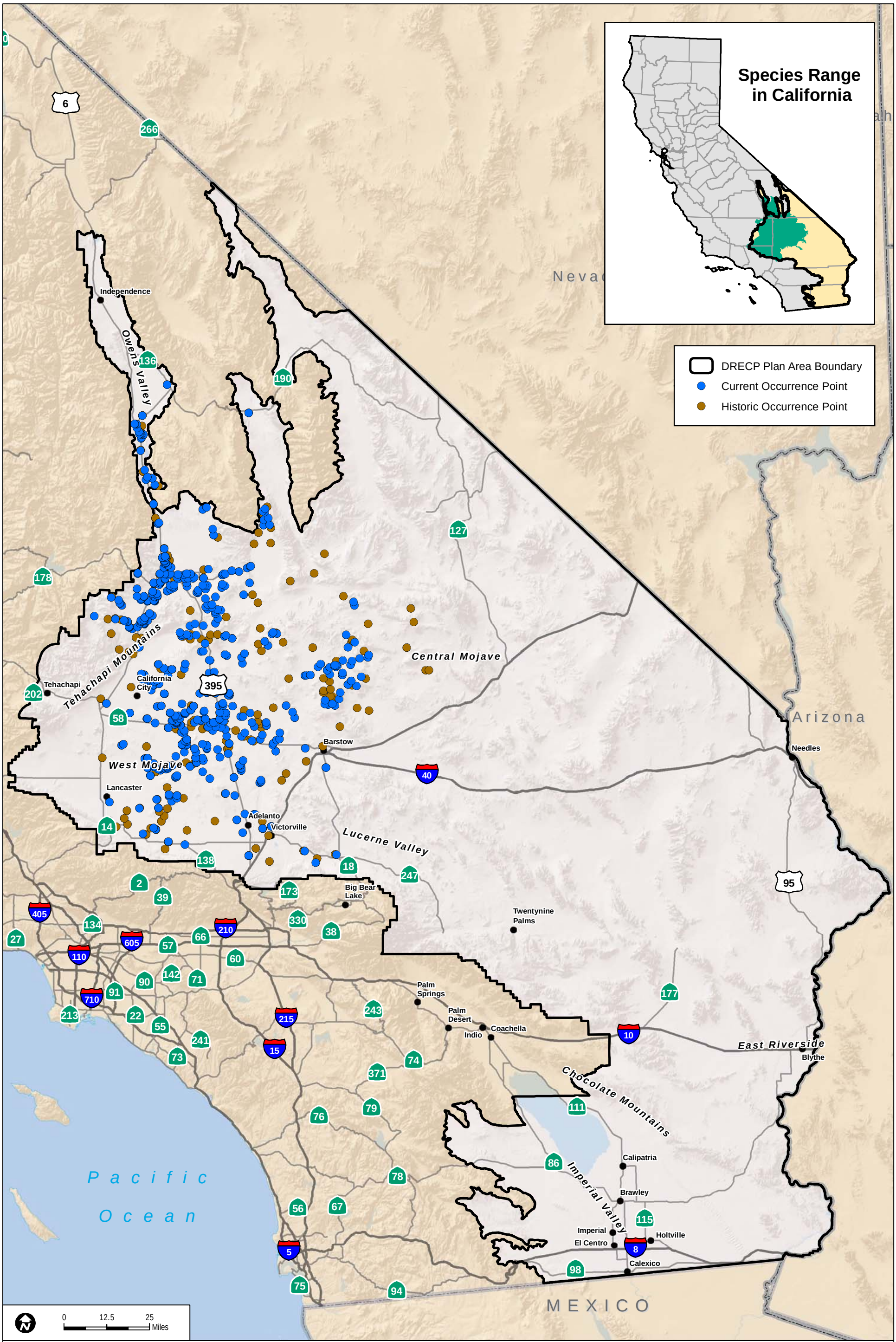

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M05

Mohave Ground Squirrel Occurrences in the Plan Area

Pallid Bat (*Antrozous pallidus*)

Photo courtesy of Scott Trageser.

Legal Status

State: Species of Special Concern
Federal: Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A
Notes: None

Taxonomy

The pallid bat (*Antrozous pallidus*) is the only species in the genus *Antrozous* of the family Vespertilionidae (Hermanson and O'Shea 1983; Hoofer et al. 2003) (*Antrozous* formerly included *A. dubiaquercus*, but this Central American species is now assigned to the genus *Bauerus* [Hermanson and O'Shea 1983]). A study of phylogenetic relationships of plecotine bats using mitochrondrial ribosomal sequences supported the placement of pallid bat as a single-species genus in the family Vespertilionidae (Hoofer et al. 2003). There are seven recognized subspecies of pallid bat (Wilson and Reeder 2005), of which *A. p. pallidus* is likely the subspecies present in the Desert Renewable Energy Conservation Plan (DRECP) project Plan Area, although *A. p. pacificus* may also occur in the western portion of the Plan Area (Hall 1981). The status of pallid bat as California Species of Special Concern is for the full species *A. pallidus*, so a subspecific assignment is not relevant to the conservation of this species in the Plan Area. No other available information indicates other important taxonomic considerations. The species' physical characteristics are described in detail in Hermanson and O'Shea (1983).

Distribution

General

The pallid bat is widespread throughout the western United States; southern British Columbia, Canada; and mainland and Baja California, Mexico (Hermanson and O'Shea 1983; Hall 1981). Within the United States, it ranges east into southern Nebraska, western Oklahoma, and western Texas (Figure SP-M07). The pallid bat is locally common in the Great Basin, Mojave, and Sonoran deserts (especially the Sonoran life zone) and grasslands throughout the western United States, and it also occurs in shrublands, woodlands, and forests at elevations up to 2,440 meters (8,000 feet) (Hermanson and O'Shea 1983; Hall 1981). The pallid bat occurs throughout California, except at the highest elevations of the Sierra Nevada range. Although this species prefers rocky outcrops, cliffs, and crevices with access to open habitats for foraging, it has been observed far from such areas (Hermanson and O'Shea 1983).

Distribution and Occurrences within the Plan Area

Historical

The DRECP database for pallid bat, composed of Bureau of Land Management (BLM) and California Natural Diversity Database (CNDDB) (CDFW 2013) records, and observations by Brown (CDFW 2013; Dudek 2013), includes 20 historical records (i.e., pre-1990) for the Plan Area, dating from 1911 to 1981, and two with an unknown observation date. An additional 11 records are from areas within 5 miles of the Plan Area boundary. The historical occurrences in the Plan Area include the southern Owens Valley–eastern Sierra Nevada–Inyo Mountains area, the Mesquite Mountains in eastern San Bernardino County, the Twentynine Palms area, the lower Colorado River, and the Salton Sea area.

See Figure SP-M07 for historical and recent occurrences of pallid bat in the Plan Area.

Recent

There are 40 recent (i.e., since 1990) records in the Plan Area and 10 additional records within the 5-mile buffer area around the Plan Area (CDFW 2013; Dudek 2013). The geographic areas of recent occurrences are similar to the historical occurrences, with small clusters of observation in the Owens Valley–eastern Sierra Nevada area, Providence Mountains, Kingston Range, Avawatz Mountains, Cady Mountains, Twentynine Palms area, Little San Bernardino Mountains, Hexie Mountains, the Lower Colorado River, Chocolate Mountains, and the Peninsular Range in east San Diego County.

As with the historical data, the specificity of these recent occurrence data is variable, with some records identifying roosts and others only including general location information for observations. This dataset, therefore, should be viewed as reflecting the recent documented distribution of the species in the Plan Area and should not be used as detailed data for specific roost sites.

Natural History

Habitat Requirements

Pallid bat day roosting habitat typically includes rocky outcrops, cliffs, and spacious crevices with access to open habitats for foraging (Hermanson and O'Shea 1983; Vaughan and O'Shea 1976). Pallid bats may also roost in caves, mines, bridges, barns, porches, and bat boxes, and even on the ground under burlap sacks, stone piles, rags, baseboards, and rocks (Beck and Rudd 1960; Rambaldini 2006). Radiotelemetry data has also shown that in the desert pallid bats will roost in holes on the ground and in rock crevices on creosote bush flats, not just in mountain ranges (Brown, pers. comm. 2012). Up to the late 1940s, they were common in buildings at low elevations of the South Coast Ecoregion (Miner and Stokes 2005). For example, in the Newhall area of Southern California, they recently were observed using buildings for both day and night roosts (Johnson 2006). In Northern California, they were observed using buildings and large-diameter, tall, live trees and snags in mature forest stands for both day and night roosting (Baker et al. 2008). In Baker et al. (2008), live trees and snags used for roosting were consistently tall in height,

large in diameter, and located in mature stands in micro-sites with low percentages of overstory and mid-story cover. Day roosts generally are warm, have obstructed entrances and exits, and are high enough to avoid terrestrial predators (Rambaldini 2006). A study of night roosts, including rock overhangs, bridges, and buildings, in Oregon found that they were protected from rain and allowed free flight space for bats in and out of the roost (Lewis 1994).

Although pallid bats may use a variety of roosting habitats, they are also selective of roost sites with microenvironments that minimize energy expenditure through adaptive hypothermia and maintain low metabolic rates (Vaughan and O'Shea 1976). In spring and fall at roost sites in Central Arizona, they used vertical crevices that passively warmed during the afternoon prior to emergence, and in the summer, they used deep horizontal crevices that acted as heat sinks and kept ambient temperatures low (Vaughan and O'Shea 1976). A roost temperature of about 30 degrees Celsius (86 degrees Fahrenheit) is considered about optimal for maintaining low metabolic rates (Trune and Slobodchikof 1976; Vaughan and O'Shea 1976). In desert regions, roost sites are often near water, although they have been observed in areas without apparent water sources (Hermanson and O'Shea 1983).

Pallid bat day roosts consisting of single- or mixed-sex colonies usually are established in crevices or man-made structures. Day roosts usually have at least 20 individuals and sometimes more than 200 individuals (Hermanson and O'Shea 1983).

Foraging habitats for pallid bats are varied and include grasslands, oak savannah woodlands, open pine forests, talus slopes, and agricultural areas (Rambaldini 2006). In a study of bat use of riparian habitats in southern Nevada, including riparian marsh, mesquite bosque, riparian woodland, and riparian shrubland, Williams et al. (2006) recorded about 88% of pallid bat occurrences in riparian woodland. Although most foraging probably occurs in close proximity to night roosts, movements greater the 2 kilometers (1.2 miles) from roosting sites in forest habitats are common (Baker et al. 2008), and movements up to 30 kilometers (18.6 miles) have been recorded (Hermanson and O'Shea 1983). See discussion in Spatial Behavior for more information.

Table 1 summarizes the likely habitat associations for pallid bat in the Plan Area.

Table 1. Habitat Associations for Pallid Bat

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Rocky, Barren, and Unvegetated Community	Day and night roosts	Day and night roosting	>50% rocky slopes within 6.2 miles of water source	Hermanson and O'Shea 1983
All natural land covers (i.e., except developed and disturbed)	Foraging	Primary foraging	Natural land covers within 3.1 miles of day roosting habitat	Baker et al. 2008; Bell 1982; Rambaldini 2006
All natural land covers (i.e., except developed and disturbed)	Foraging	Secondary foraging	Natural land Covers 3.1 to 6.2 miles of day roosting habitat	Baker et al. 2008; Bell 1982; Rambaldini 2006

Notes: Water sources include major rivers, reservoirs, lakes, ponds, seeps and springs, and perennial streams. Pallid bats are expected to forage in virtually all relatively open, natural land covers in the Plan Area where suitable prey are present.

Foraging Requirements

Pallid bats forage about 0.5 to 2.5 meters [1.6 to 8.2 feet] above the ground surface, and their foraging behavior is directed toward prey that are close to the ground, on the ground, or perched on exposed vegetation (O'Shea and Vaughan 1977). They may forage both aerially and by gleaning from plants, and they have also been observed to take prey by crawling along the ground. Their diet generally has been described to include scorpions, ground crickets, solpugids, darkling ground beetles, carrion beetles, short-horned grasshoppers, cicadas,

praying mantids, long-horned beetles, and sphingid moths (Hermanson and O'Shea 1983). While pallid bats are primarily insectivores, they have also been observed to eat lizards and smaller bats in captivity (Hermanson and O'Shea 1983) and likely take a variety of small vertebrates in the wild. Their specific diets vary geographically and may reflect genotypic or phenotypic selection (Johnston and Fenton 2001). Pallid bats generally take large prey (up to 6.0 centimeters [2.4 inches] total body length) (O'Shea and Vaughan 1977). In both a coastal area (Marin County) and a desert area (Caliente Mine in Death Valley) in California, pallid bats foraged for Orthoptera (grasshoppers, crickets) and Coleoptera (beetles), and smaller percentages of Solpugida (sun scorpions), Lepidoptera (moths), and Diptera (flies). At Caliente Mine, Coleoptera made up about 55% of their diet by volume, but diet changed over time, reflecting the availability of prey. Individuals in the local population tended to have the same diet at any given time (Johnston and Fenton 2001). In contrast, at the Marin County site, diets were varied, but the variation was related to individual differences (i.e., there was no "average" diet for the group such as that of the Caliente site), and these differences may have reflected learning that reduces searching and handling time (Johnston and Fenton 2001).

Reproduction

Pallid bats breed in October through December, and possibly through February (Hermanson and O'Shea 1983) (Table 2). Females store sperm and ovulation occurs during the following spring. Gestation is approximately 9 weeks, and birth in the southwestern United States typically occurs from May through June (Hermanson and O'Shea 1983). Litter size is typically 2 young (approximately 80% of litters (Bassett 1984)), and occasionally 3; yearling females may breed but litter size is 1 (Davis 1969; Hermanson and O'Shea 1983). The young are born relatively undeveloped, but they mature rapidly and engage in their first flight at 33 to 36 days (Davis 1969). They achieve full adult flight capability by about 49 days of age and full adult weight by 56 days of age (Hermanson and O'Shea 1983). Yearling males are not sexually active their first autumn and probably not their first year (Davis 1969). Mature males and females have the same body

dimensions (e.g., weight, forearm length, wing area); they do not exhibit sexual dimorphism (Davis 1969).

Pallid bats have lived up to 9 years in captivity (Hermanson and O'Shea 1983).

Table 2. Key Seasonal Periods for Pallid Bat

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding	?	?								x	x	x
Birth/ Development					x	x	x	x				
Winter Torpor	x	x	x	x								x

Sources: Bassett 1984; Davis 1969; Hermanson and O'Shea 1983.

Spatial Behavior

Pallid bats in central Arizona exhibited a bimodal foraging activity pattern, with two foraging bouts separated by a period of night roosting, with the timing and duration of these activities seasonally variable (O'Shea and Vaughan 1977). During the summer months, time away from the roost varies between approximately 45% to 58% of the night. In September and October, time away from the roost varies between 25% to 27% of the night. (O'Shea and Vaughan 1977) Pallid bats may be active outside the roost any time of year, but their activity during the winter may be erratic, which probably is associated with cold periods when they are in torpor (Table 2). They have been mist-netted at temperatures as low as 2 degrees Centigrade (35.6 degrees Fahrenheit) in southern Nevada (O'Farrell et al. 1967). In contrast to O'Shea and Vaughan (1977), O'Farrell et al. (1967) did not detect a bimodal activity period in southern Nevada during the fall and winter; all captures were 1.5 to 5 hours after dusk. This information indicates that nightly foraging activity by pallid bats is seasonally variable.

During July through August, pallid bats in central Arizona showed little fidelity to specific roosting sites, but during the cooler months

they showed greater fidelity to certain roosting sites (O'Shea and Vaughan 1977), which may reflect more specific roost requirements during the colder months to maintain thermoregulation (also see Habitat Requirements regarding day roost characteristics).

The distances that pallid bats travel during foraging bouts may be limited by the availability of night roosts because they frequently bring large prey to these sites where it is then eaten (O'Shea and Vaughan 1977). Bell (1982), for example, observed pallid bats foraging within 3 kilometers (1.9 miles) of roost sites in desert grasslands in New Mexico. A radio-tracking study in British Columbia found that foraging occurred within 1.5 kilometers (0.9 mile) of day roost sites (Rambaldini 2006). In this study, males returned to the day roost for short periods between foraging bouts (Rambaldini 2006) (however, note from discussion above that nightly foraging activity is seasonally variable). In coniferous forest in Northern California, radio-tracking documented that foraging bouts more than 2 kilometers (1.2 miles) from the day roost were common, but most foraging occurred in close proximity to day roosts (Baker et al. 2008). The longest distance moved during this study was 4.7 kilometers (2.9 miles) by a pregnant female. Lactating females had average foraging ranges of 1.56 square kilometers (0.6 square mile), and post-lactating females had average ranges of 5.97 square kilometers (2.3 square miles) (Baker et al. 2008). However, flights up to 30 kilometers (19 miles) between night roosts have been recorded, indicating that pallid bats have the capacity to fly long distances. Further, homing studies have shown a maximum return distance of 174 kilometers (108 miles), and several recoveries have shown return distances of 48 to 51 kilometers (30 to 32 miles) from release sites within 7 to 8 hours after release (Hermanson and O'Shea 1983).

Dispersal flights in the central Arizona study occurred in mid-August and were characterized by straight-line flight movements from the day roost (in contrast to the typical circling of the roost area) at approximately 25 meters (82 feet) above the ground and no evidence of foraging (O'Shea and Vaughan 1977). These dispersal flights occurred at the same time the population numbers at the day roost sharply declined (O'Shea and Vaughan 1977), indicating that young were leaving the maternity site.

Ecological Relationships

Day roost selection, fidelity, and lability (flexibility) by pallid bats indicate potentially important ecological relationships and are region-specific. As discussed in Habitat Requirements, pallid bats select day roosts that appear to maximize adaptive hypothermia (Vaughan and O'Shea 1976). In addition to microclimate stability, deep crevices used for day roosts may provide protection from predators and protection of juveniles that may fall from the ceiling (Lewis 1995). In central Arizona, where such deep crevices are available, females change day roosts in the spring, but not during pregnancy and lactation (O'Shea and Vaughan 1977). In Oregon, where such deep crevices are not available for roosting, females change day roosts throughout the summer (Lewis 1995). Lewis (1995) suggests that the Oregon populations benefit from roost lability by reducing ectoparasite infestations. In Arizona, the benefits of roost fidelity to the deep crevices may outweigh the impacts of ectoparasites (Lewis 1995).

In addition to selecting roosting sites to maximize adaptive hypothermia, social roosting also appears to be important for conserving metabolism. An experimental study showed that individual roosting bats had higher metabolic rates and weight loss than bats roosting in clusters and at suboptimal temperatures of 25 and 35 degrees Celsius (77 and 95 degrees Fahrenheit) (Trune and Slobodchikoff 1976).

Pallid bats may share both day and night roosts with other bat species such as Brazilian free-tailed bat (*Tadarida brasiliensis*) and Yuma myotis (*Myotis yumanensis*)(Hermanson and O'Shea 1983; Licht and Leitner 1967), but there is no evidence in the literature of competitive or symbiotic relationships with other bats. Congregations with other bat species at both day and night roosts may simply reflect use of limited resources.

Black (1974) suggested that bats may employ several types of foraging and food partitioning mechanisms that could reduce inter-specific competition, including size and type of prey; periods of activity (most bat prey are active within a few hours of sunset, but different prey have different peak activity periods); spatial partitioning, such as between-,

within-, and below-canopy foragers; and flight patterns, such as slow vs. fast flying, maneuverability, and hovering.

Compared to other bat species, pallid bats emerge from day roosts relatively late in the evening (Hermanson and O'Shea 1983), but there is no information to suggest that this reflects competition for prey with other species. Artificial lighting may affect competitive predator-prey relationships among bats. Longcore and Rich (2004) suggest that artificial lighting, which attracts many insects taken by bats, including moths (Frank 1988), may alter local community relationships because the faster-flying bats congregate around lights and can exploit this concentrated food source while slower-flying bats avoid lights and are unable to benefit from this concentration of insects; however, whether this applies to pallid bats, which tend to concentrate their foraging near or on the ground, is unknown.

Colony sizes are variable, but maximum densities appear to be related to mid-summer densities of insect prey (Hermanson and O'Shea 1983). As discussed previously in Foraging Requirements, pallid bats often feed on ground insects, which may make them more vulnerable to injury and predation (Hermanson and O'Shea 1983).

Population Status and Trends

Global: Secure (NatureServe 2011)
State: Vulnerable (CDFW 2013)
Within Plan Area: Same as state

Pallid bat is a California Species of Special Concern, but little data is available to assess population status and trends. Ellison et al. (2003) compiled 292 observations for 133 colonies in 11 western states, including 35 (12%) from California. About 35% of the observations were from Arizona, 18% from Oregon, and 10% from New Mexico. However, most (78%) of the observations were collected before 1990. Information from only two sites was adequate to assess population trends: a bridge roost in Arizona that declined from 80 individuals to zero and a decline in a colony using crevices in cliffs in the Verde Valley of Arizona concurrent with increases in human activity in the area (Ellison et al. 2003). In California, Miner and Stokes (2005) noted a serious decline of pallid bats in the South Coast Ecoregion, especially

in low-lying areas. They report that even as late as 1948 the species was considered to be abundant in buildings, but that by the 1970s only 1 of 12 known roost sites was still extant. Recent survey information for San Diego County indicates that few roosts that support bat species typically found in association with the pallid bat also include the species (Miner and Stokes 2005). Based on this apparent population decline, Miner and Stokes (2005) concluded that pallid bats are highly intolerant of urban development.

Threats and Environmental Stressors

As a colonial roosting species, pallid bats are particularly vulnerable to disturbances of roost sites through vandalism, extermination, and destruction of buildings used as roost sites (Hermanson and O'Shea 1983), as well as to recreational activities such as rock climbing. As noted previously, a decline in an Arizona colony occurred concurrent with an increase in human activity (Ellison et al. 2003). Miner and Stokes (2005) found that pallid bats have abandoned almost all previously occupied sites in the urbanized areas of the South Coast Region since the late 1940s. Beck and Rudd (1960) observed that female pallid bats are particularly sensitive to disturbance during the period prior to giving birth through weaning. A single disturbance may cause them to abandon the maternity roost prior to giving birth or to move to a more secluded part of the roost after giving birth (Beck and Rudd 1960).

Food availability may be reduced by pesticides or habitat modification or degradation such as conversion to agriculture, prescribed fires, and wildfires. Pesticides and heavy metals also may contaminate prey, causing secondary poisoning. Because this species often forages on the ground, it is susceptible to predation by urban-related predators (e.g., cats and possibly dogs) and potentially collection or harassment by humans.

Several recent studies have documented substantial mortality of bats at wind energy facilities (e.g., Baerwald and Barclay 2009; Cryan 2011; Cryan and Barclay 2009). While, as of 2010, there have been no reported fatalities of pallid bats at wind energy facilities (e.g., Tetra Tech EC Inc. 2010), Solick and Erickson (2009) indicate that there have been relatively few systematic, post-project, bat-fatality monitoring data collected for large, wind-energy projects in the arid southwestern United States. Although fatalities of this species at wind

energy facilities have not been documented, it is expected that the species could be at risk from turbine strikes, or other factors associated with turbine operation, such as barotrauma, hypothesized to cause bat fatalities at wind facilities (Cryan and Barclay 2009). Pallid bats would be at greatest risk of turbine strikes or from other associated causes if a facility was located within a few miles of a day roost site (where most foraging activity occurs), and strikes would most likely occur during emergence and return to the day roost. Risk of strikes may also be higher during dispersal when young are leaving the natal roost site and fly in straight lines from the roost at altitudes of 80 feet or more (O'Shea and Vaughan 1977). Risk of strikes may be relatively low during foraging activities because pallid bats tend to forage on or close to the ground.

Conservation and Management Activities

Pallid bat is addressed in the West Mojave Plan (BLM 2005). Under Alternative A (the Proposed Action – Habitat Conservation Plan), BLM would implement several conservation measures for pallid bat, including:

- Protection of all significant roosts (defined as maternity and hibernation roosts supporting 10 or more individuals) by installing gates over mine entrances and restricting human access (The West Mojave Plan identified two significant maternity roosts and one significant maternity/hibernation roost for pallid bat on BLM-managed lands);
- Protection of bat roosts in the Pinto Mountains by gating known and new significant roosts and notifying claim holders on BLM lands containing significant roosts;
- Continued fencing around (but not over) open, abandoned mine features to provide bats access to roosts and to reduce hazards to the public;
- Required surveys for bats by applicants seeking discretionary permits for projects that would disturb natural caves, cliff faces, mine features, abandoned buildings, or bridges to determine whether significant roost sites are present; and
- Safe eviction of bats at a non-significant roost (i.e., fewer than 10 individuals) prior to disturbance or removal.

In addition, as a BLM sensitive species, pallid bat is addressed under land use actions undertaken by BLM. In accordance with BLM's "6840 – Special Status Species Management" manual, the objectives for sensitive species policy are:

> To initiate proactive conservation measures that reduce or eliminate threats to Bureau sensitive species to minimize the likelihood of and need for listing of these species under the ESA" (BLM 2008).

Under this policy, BLM must consider the impact of actions on sensitive species, including outcomes of actions (e.g., land use plans, permits), strategies, restoration opportunities, use restrictions, and management actions necessary to conserve BLM sensitive species.

Pallid bat is also addressed in the Military Integrated Resource Management Plans (INRMP) for the China Lake Naval Air Weapons Station (NAWS and BLM 2004) and the Marine Air Ground Task Force Training Command, Marine Corps Air Ground Combat Center, Twentynine Palms (MAGTFTC MCAGCC 2007). As a designated sensitive species in these INRMPs, pallid bat is provided protection and management considerations during the land use planning process defined in the China Lake Comprehensive Land Use Management Plan and military training operations at Twentynine Palms. If it is determined to be at risk from a proposed project or training activities, efforts are made to avoid and minimize impacts. For example, at Twentynine Palms, four bat gates have been installed in three mines to allow bats access to roosts without disturbance from humans. The Twentynine Palms INRMP also includes three objectives:

- Monitoring current bat gates to inspect for trespass and condition;
- Evaluating mine entrances for installation of bat gates to those mines that are exceptional bat habitat but not culturally significant; and
- Evaluating modification of bighorn sheep guzzlers for use by bats and other wildlife to enhance habitat value.

Data Characterization

There are relatively few data for pallid bat in the Plan Area. As noted in Distribution and Occurrences with the Plan Area, there are only 59 data records for the Plan Area, of which 39 are recent. Although this species is considered common in the Great Basin, Mojave, and Sonoran deserts, there is little information about roost sites, particularly winter roosting sites and hibernacula. There is also little information on seasonal movements.

Management and Monitoring Considerations

The primary management and monitoring consideration for the pallid bat is protection of day and night roosts from disturbance that may cause abandonment. This species requires very specific thermal conditions in day roosts (e.g., deep crevices that provide an optimum thermal environment), plus the additional factor that day roosts tend to be near water resources. These habitat requirements likely result in relatively few highly suitable day roosting sites in the Plan Area. Any occupied day roosts, therefore, should be considered a highly valuable resource, and impacts should be avoided. Maintaining these sites will require protecting them from human disturbances and adjacent land uses that could cause direct mortality or injury of pallid bats or abandonment of the roost site.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for pallid bat, using available spatial information and occurrence information, as appropriate. For this reason, the term “modeled suitable habitat” is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 19,196,457 acres of modeled suitable habitat for pallid bat in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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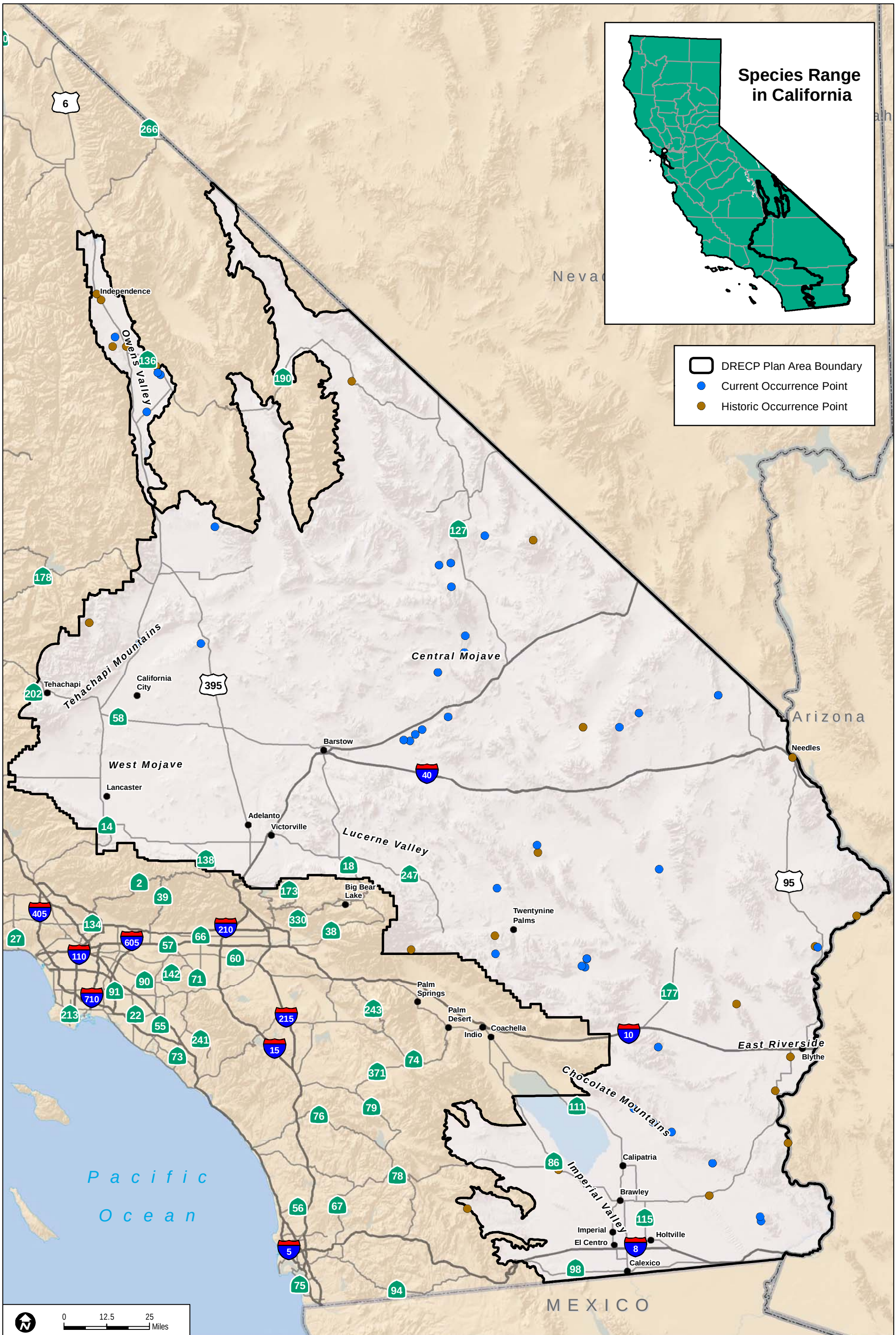

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M06

Pallid Bat Occurrences in the Plan Area

Townsend's Big-Eared Bat (*Corynorhinus townsendii*)

Photo courtesy of Rob Schell Photography.

Legal Status

State: Species of Special Concern
Federal: Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

The taxonomy of Townsend's big-eared bat (*Corynorhinus townsendii*) has undergone some recent revisions. Although the species was originally assigned to the genus *Corynorhinus* (Hall 1981), Handley (1959) reassigned it to the genus *Plecotus*, based on physical measurements, with *Corynorhinus* placed in a subgenus. More recent phylogenetic work using physical characters (Frost and Timm 1992; Tumlison and Douglas 1992) and mitochondrial DNA analysis (Hoofer and Van Den Bussche 2001) have resulted in *Corynorhinus* being restored to a separate genus within the plecotine bats.

There has also been past uncertainty in California about the distinction and distributions of two subspecies: *C. t. townsendii* and *C. t. pallescens* (see discussion in CDFG 1998). While the two subspecies occur in geographically discrete locations, their distributions have been recently revised based on mitochondrial DNA, with C*. t. townsendii* occurring throughout western and southwestern Canada and *C. t. pallescens* generally limited to New Mexico and Colorado (Piaggio et al. 2009). There are areas of sympatry in Colorado where the two subspecies are not genetically different (Piaggio et al. 2009), but based on genetic information, the subspecies in California and the Desert Renewable Energy Conservation Plan (DRECP) Area is *C. t. townsendii*. Nonetheless, in California the full species *Corynorhinus townsendii* is designated a Species of Special Concern, so the subspecific distinction in the distribution of *C. t. townsendii* and *C. t. pallescens* is not critically important for planning purposes. The species' physical characteristics are described in detail in Kunz and Martin (1982).

Distribution

General

The Townsend's big-eared bat ranges throughout the western United States; British Columbia, Canada; and Mexico (Kunz and Martin 1982). In the United States, it occurs in a continuous distribution in all of the western states and east into western South Dakota, northwestern Nebraska, southwestern Kansas, western Oklahoma, and western Texas (Piaggio et al. 2009). This continuous distribution comprises three subspecies: *C. t. townsendii,* which based on the recent genetic data (Piaggio et al. 2009) has the largest distribution range from Canada south into Mexico; *C. t. pallescens,* which is primarily limited to Colorado and New Mexico; and *C. t. australis*, which occurs in southwestern Kansas, western Oklahoma, western Texas, and north–central Mexico (Piaggio et al. 2009). The other two subspecies occur in disjunct distributions: *C. t. ingens* in southeastern Kansas, northeastern Oklahoma, southwestern Missouri, and northwestern Arkansas; and *C. t. virginianus* in eastern Kentucky, West Virginia, and Virginia (Piaggio et al. 2009).

Within California, Townsend's big-eared bat occurs throughout the state, with the exception of alpine and subalpine areas of the Sierra Nevada (Figure SP-M08), although they have been found in the subalpine zone in the White Mountains to the east of the Sierra (Szewczak et al. 1998).

Distribution and Occurrences within the Plan Area

Historical

Townsend's big-eared bat may occur throughout the Plan Area, but there are relatively few documented large maternity and/or hibernation roosts. A comprehensive review of the species' distribution was conducted by Pierson and Rainey (CDFG 1998) based on a review of historical records and field surveys conducted from June 1987 to January 1991. Their review included portions of the Plan Area known to support substantial populations, including the Owens Valley and areas east of the Sierra Nevada Range in Inyo County, the Providence Mountains in San Bernardino County, and the lower

Colorado River area in San Bernardino, Riverside, and Imperial counties (see Figure 1 in CDFG 1998). They surveyed all known maternity colonies with at least 30 individuals. Most of the active large maternity roosts within or near the Plan Area were in abandoned mines east of the Sierra Nevada range and the western slopes of the White Mountains bordering the Owens Valley. Active maternity roots were also found in the Kingston Range area of eastern Inyo County, the Providence Mountains in northeastern San Bernardino County, and along the lower Colorado River in eastern Riverside County. An active maternity roost and a hibernation roost were also found in east San Diego County. No longer active roosts (i.e., previously known roost sites) or roosts made unavailable by human activities (e.g., inappropriate gating) were found in the Coso Range area of southern Inyo County, a site in the Providence Mountains, and two sites along the Lower Colorado River in Riverside and Imperial counties, respectively (see Figure 1 of CDFG 1998). As of 1991, Pierson and Rainey (CDFG 1998) estimated 11 active sites east of the Sierra Nevada (including several sites north of the Plan Area and the site in the Kingston Range) totaling about 1,300 adult females, 1 site in the high desert totaling about 75 adult females, 1 site in the lower desert totaling about 50 adult females, and the 2 east San Diego County sites with an unknown number of adult females. Pierson and Rainey (CDFG 1998) indicate that no large hibernation sites have been found in the desert regions of California and that smaller hibernation sites (5 to 20 individuals) are more typical of the desert; these sites are not included in the data reported by Pierson and Rainey. The lack of documented large hibernation sites in the Plan Area may reflect a lack of extensive exploration of mines and caves at higher elevations where they would more likely hibernate (CDFG 1998). However, because it is unlikely that mines and caves in the Plan Area, which are at lower elevations, have subsurface temperatures low enough for hibernation (i.e., less than 10 degrees Celsius [50.0 degrees Fahrenheit]) (see discussion in Habitat Requirement), additional exploration for hibernation sites may be irrelevant (Szewczak, pers. comm. 2012).

The DRECP database for Townsend's big-eared bat, comprising Bureau of Land Management (BLM) and California Natural Diversity Database (CNDDB) (CDFW 2013; Dudek 2013) records, includes 13

historical records (pre-1990) for the Plan Area, dating from 1914 to 1983, as well as one record with an unknown observation date. An additional 8 records are from areas within 5 miles of the Plan Area boundary. These data generally accord with the information provided in Pierson and Rainey (CDFG 1998), with clusters of occurrences in the southern Owens Valley–eastern Sierra Nevada area, especially the mountain ranges north of Ridgecrest. Historical records are also known from the Providence Mountains, the Kingston Range, the lower Colorado River, and Hesperia north of the San Bernardino Mountains.

See Figure SP-M08 for current and historical occurrences of Townsend's big-eared bat in the Plan Area.

Recent

There are 39 recent (i.e., since 1990) records in the Plan Area and 42 additional records within the 5-mile buffer area around the Plan Area (CDFW 2013; Dudek 2013). The geographic areas of the recent occurrences are similar to the historical occurrences, with clusters of observations in the Owens Valley–eastern Sierra Nevada area, Providence Mountains, and the Kingston Range. There is also a cluster of recent occurrences north of Barstow and along the northern slopes of the San Bernardino Mountains. There are relatively few recent occurrences from the lower Colorado River, consistent with the information reported by Pierson and Rainey (CDFG 1998).

As with the historical data, the specificity of these recent occurrence data is variable, with some records identifying roosts and others only including general location information for observations. This dataset, therefore, should be viewed as reflecting the recent documented distribution of the species in the Plan Area and should not be used as detailed data for specific roosts sites.

Natural History

Habitat Requirements

Townsend's big-eared bat is primarily associated with mesic habitats characterized by coniferous and deciduous forests and riparian habitat, although it also occurs in xeric areas (Kunz and Martin 1982). In

California, this species was historically associated with limestone caves and lava tubes located in coastal lowlands, agricultural valleys, and hillsides with mixed vegetation. The species also occurs in man-made structures and tunnels (Kunz and Martin 1982), mines (López-González and Torres-Morales 2004), and the basal hollows of old-growth redwood trees (*Sequoia sempervirens*) on the north coast of California (Gellman and Zielinski 1996; Zielinski and Gellman 1999). Within the Plan Area, Townsend's big-eared bat is primarily associated with mines in the California desert and also largely associated with man-made structures, tunnels, caves, and the basal hollows of old-growth redwood trees. In a study in northern Utah, caves and mines were the most frequently used type of roosts. More than 84% of roosts were in caves, and more than 21% of abandoned mines were used as day roosts; notably, no bridges were used (Sherwin et al. 2000). Occupied day roosts typically were subject to little disturbance by humans. Maternity colonies tended to be located in large complex sites with multiple openings (Sherwin et al. 2000). It has been suggested that the Townsend's big-eared bat has become more common in the western United States due to the availability of man-made structures (Kunz and Martin 1982);however, see discussion under Population Status and Trends. Many roosting sites in the California coastal area are in buildings, but in the Plan Area most roosting sites appear to be in abandoned mines (CDFG 1998).

Unlike many cave-roosting bat species, Townsend's big-eared bat only roosts in the open, often hanging from walls and ceilings (CDFG 1998). In the summer maternity roosts, females roost in the warm parts of caves and buildings in clusters (Kunz and Martin 1982). The census of maternity roosts in California found an overall mean colony size of about 112 individuals (CDFG 1998), which is larger than generally reported in the literature (e.g., Kunz and Martin 1982). Males appear to roost solitarily near the maternity roosts. In winter, roosting occurs solitarily or in small clusters, and Townsend's big-eared bat may share hibernacula with other bat species (Kunz and Martin 1982) (see Ecological Relationships). This species may require relatively cold temperatures to hibernate (Humphrey and Kunz 1976). Townsend's big-eared bats roost in relatively cold parts of caves in well-ventilated areas near entrances, but may move to more temperate parts of the cave if temperatures become too cold (e.g., subfreezing) (Clark et al.

2002; Humphrey and Kunz 1976; Kunz and Martin 1982) (also see discussion under Spatial Activity).

Pierson and Rainey (CDFG 1998) provide detailed information for the physical features of roosting sites in California, which is summarized below. The reader is directed to the Pierson and Rainey report for more detailed information.

Pierson and Rainey (CDFG 1998) examined potentially suitable and accessible caves, tunnels (e.g., old mine workings, water diversion tunnels, and abandoned railroad tunnels), abandoned and little-used buildings, and older (pre-1960) bridges throughout California. Censuses of bats at occupied roosts were based on direct counts or estimates for an area covered by a cluster of bats. The physical characteristics of roosts described as follows are summarized from Pierson and Rainey (CDFG 1998).

As of 1998, maternity roosts were distributed among the different structures as follows: 23 (43%) in caves; 21 (39%) in mines; 8 (15%) in buildings; and 2 (4%) in other structures (an abandoned bridge and a diversion tunnel). All roosts could be classified structurally as "cave analogues" that contained a relatively large, but enclosed space with a substantial opening. All but one of the roost entrances ranged from at least 15 centimeters (5.9 inches) in height and 31 centimeters (12.2 inches) in width, with the smallest being 15 centimeters (5.9 inches) high and 46 centimeters (18.1 inches) wide. The one exception was a mine roost in which the opening was about 10 centimeters (3.9 inches) high and 60 centimeters (23.6 inches) wide. All roosting sites were at least 1 meter (3.3 feet), and usually 2.5 to 5.0 meters (8.2 to 16.4 feet) off the ground. All roost sites were classified as semi-dark to dark settings. Mean temperatures of maternity roosts and roosts occupied by single individuals and small clusters were not significantly different. The mean temperature of maternity sites was 24.1 degrees Celsius (75.4 degrees Fahrenheit), and the mean temperature of sites with individuals and small clusters was 22.2 degrees Celsius (72.0 degrees Fahrenheit). The temperature range for maternity sites was typically 18 to 30 degrees Celsius (64.4 to 86.0 degrees Fahrenheit), but was measured as low as 14 degrees Celsius (52.2 degrees Fahrenheit). Roost relatively humidity was not a factor, but tended to be relatively dry on average at about 33% (range 19 to 93%).

Assessing and characterizing hibernacula was more difficult than maternity sites because individuals tend to move among different sites during a hibernation season (CDFG 1998; Sherwin et al. 2003). Similar to maternity roosts, hibernacula are typically caves, or cave analogues, but differ in often being L-shaped, with vertical and horizontal entrances that generate a "cold sink" with significant air flow. Consistent with the literature for the species, hibernacula used in California often represent the coldest non-freezing temperature available. In the northern counties of Shasta, Siskiyou, and Lassen, where individuals probably hibernate longer periods of time, mean hibernating roost temperature was 4.3 degrees Celsius (39.7 degrees Fahrenheit). In warmer regions of coastal and Southern California, individuals arouse periodically during the winter and occur in warmer hibernacula. The mean hibernaculum temperature for known sites throughout California is 7.1 degrees Celsius (44.8 degrees Fahrenheit)), and preferred hibernating temperatures are always below 10 degrees Celsius (50.0 degrees Fahrenheit) (CDFG 1998).

Townsend's big-eared bats forage for insects in a variety of habitats, primarily between the canopy and mid-canopy of forests, woodlands, and riparian zones, but also in sagebrush shrubsteppe (Fellers and Pierson 2002). Fellers and Pierson (2002) noted that Townsend's big-eared bats avoided foraging in grasslands. As discussed below in Spatial Activity, most foraging occurs in relatively close proximity to the day roost.

Potential roosting and foraging habitat associations for Townsend's big-eared bat in the Plan Area are provided on Table 1.

Table 1. Habitat Associations for Townsend's Big-Eared Bat

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Abandoned mines	Day roosts	TBA	CDFG 1998
Woodland, forest, riparian, desert wash	Foraging	Woodland, forest, riparian, desert wash within 6.2 miles of day roosting habitat	Fellers and Pierson 2002

Foraging Requirements

Several studies in various parts of the Townsend's big-eared bat's range found that Lepidoptera (moths) are its primary prey, including in the southwest (Ross 1967), eastern and western Oregon (Whitaker et al., 1977, 1981), and Virginia (Sample and Whitmore 1993). In Oregon, big-eared bats feed almost exclusively on moths (Whitaker et al. 1977, 1981). In Virginia, moths comprised about 90% of the species' diet by volume and percentage, followed by Coleoptera (beetles), Diptera (flies), and Hymenoptera (bees and wasps), and reflected the abundance of these orders in interior forests (Sample and Whitmore 1993).

Reproduction

Reproduction by Townsend's big-eared bats in California is fairly well known, based on a study by Pearson et al. (1952), described herein (Table 2). Breeding begins in autumn, with peak breeding in November through February. Females store the sperm until ovulation in the spring, which may occur during and after females leave hibernation. Upon leaving hibernation, females form maternity colonies in the late spring and early summer; males during this period appear to roost singly (CDFG 1998). Gestation varies from 8 to 14 weeks, depending on degree of torpor and spring temperatures. Females have one pup. In California, birth occurs in the late spring to early summer over a 3- to 5-week period beginning in late May. Although young are born fairly undeveloped, they grow rapidly and reach adult body proportions (i.e., forearm length) in 1 month. They are capable of flying in 2.5 to 3 weeks and are weaned by 6 weeks. Both males and females are reproductive in their first autumn. Immediate postnatal mortality is about 4% to 5%, and 3-year survival is 70% to 80% for adults and 38% to 40% for yearlings (i.e., survival increases with age) (Kunz and Martin 1982).

Female maternity groups are stable and faithful to roost sites that may be used by several generations (CDFG 1998). Females remain in the natal group while males disperse after their first summer (CDFG 1998). Maternity roosts begin to break up in August.

Table 2. Key Seasonal Periods for Townsend's Big-Eared Bat

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding	x	x								x	x	x
Birth/ Development				x	x	x	x	x				
Male Dispersal								x	x			
Hibernacula	x	x	x	x	x					x	x	x

Source: Pearson et al. 1952.

Spatial Activity

Pierson and Rainey (CDFG 1998) characterize Townsend's big-eared bat as "quite sedentary" because marked animals (all females) moved no more than a few kilometers from their natal roost. Also, most activity outside of day roosts (e.g., foraging, night roosting) occurring relatively close to the roost (CDFG 1998). Recorded maximum distance from the day roost in California is 32.2 kilometers (20.0 miles) and 64.4 kilometers (39.9 miles) in Kentucky (Kunz and Martin 1982). Average distance from maternity roosts to winter hibernacula is 11.6 kilometers (7.2 miles) (range: 3.1 to 39.7 kilometers [1.9 to 24.6 miles]) (Kunz and Martin 1982). Based on a personal communication from Pearson, Pierson and Rainey (CDFG 1998) noted that when maternity colonies disband in the fall, a banded individual had never been recorded at hibernacula more than 43 kilometers (27 miles) from the banding site. However, there is also indirect evidence that Townsend's big-eared bats can travel much longer distances than indicated by direct observations of foraging activity and movement between maternity roosts and hibernacula, based on telemetry and banding studies. The genetic work by Piaggio et al. (2009) indicated gene flow by dispersing males in Colorado has occurred between roost sites 310 kilometers (192 miles) apart.

Nightly movements for bats in Marin County, California, were monitored using radiotelemetry by Fellers and Pierson (2002). Bats

typically traveled less than 10.5 kilometers (6.5 miles) from the day roost, and most flight was in the immediate vicinity of native vegetation where foraging was assumed to occur, and particularly along the edges of riparian vegetation. Similarly, on Santa Cruz Island off the coast of California, foraging activity occurred in native forest habitat within 5 kilometers (3.1 miles) of the day roost (Brown et al. 1994). Nightly foraging tended to occur in the same areas at the Marin County site (Fellers and Pierson 2002), but a study in Oregon shows shifts in foraging areas over time related to changes in prey availability (Dobkin et al. 1995). Clark et al. (1993) found that Ozark big-eared bats (*C. t. ingens*) selected foraging habitats non-randomly in relation to their availability, with edge habitats along streams and on mountain slopes used more frequently. In the Marin County study, females generally traveled greater distances than males for foraging, with their centers of activity 3.2 ±0.5 kilometers (2.0 ±0.3 miles) from the roost, compared to 1.3 ±0.2 kilometers (1.1 ±0.1 miles) for males (Fellers and Pierson 2002). Fellers and Pierson (2002) note, however, that commuting distances and patterns of nighttime activity are likely to be quite variable in relation to factors such as individual differences, sex, season, reproductive condition, and available suitable foraging habitat. For example, females may travel farther from the maternity roost or be more active foraging away from the roost later in the reproductive season when young are more independent and resources are needed to support lactation. Clark et al. (1993, 2002) found that Ozark big-eared bat nightly activity changed relative to birth and maturation of young, with nighttime returns to the maternity roost more frequent when young were totally dependent on the mother, and farther foraging distances by adult females as young matured.

Although fidelity to maternity roosts is high, there may be little fidelity to roost sites at other times of the year, possibly in relation to availability. In Oregon, there was little fidelity to night roosts in the period between emergence from hibernacula and use of maternity sites, possibly because in this study area the lava flow topography provided numerous roost sites (Dobkin et al. 1995). It is expected that use of different roost sites is locally variable in relation to roost availability.

Townsend's big-eared bats are considered to be a hover-gleaner forager based on wing morphology (Norberg and Payner 1987, as cited in Fellers and Pierson 2002), and they are agile and maneuverable

fliers. They have low wing loading and high lift capacity (Kunz and Martin 1982). Fellers and Pierson (2002) found that most flight was at 10 to 30 meters (33 to 98 feet) above ground between the mid-canopy and canopy of trees. Flight through grassland was fast and low to the ground, indicating that bats were not foraging in grasslands.

Spatial activity within roosts sites likely reflects behavioral thermoregulatory adjustments. During hibernation, individuals arouse frequently and change position or move to more temperate areas of the hibernaculum (Kunz and Martin 1982). Disturbances may also cause movements within roosts sites.

Ecological Relationships

Townsend's big-eared bats may share hibernacula with other bat species; in the eastern United States, it has been found in association with Rafinesque's big-eared bat (*C. rafinesquii*) and in the western United States with big brown bat (*Eptesicus fuscus*), cave myotis (*Myotis velifer*), western small-footed myotis (*M. ciliolabrum*), dark nosed small-footed myotis (*M. melanorhinus*),[1] and California myotis (*M. californicus*) (Kunz and Martin 1982), but there is no evidence in the literature of direct competitive or symbiotic relationships with other bats. Congregations with other bat species at both day and night roosts may simply reflect use of limited resources.

With regard to potential resource partitioning, Black (1974) suggested that bats may employ several types of foraging and food partitioning mechanisms that could reduce inter-specific competition, including size and type of prey; periods of activity (most bat prey are active within a few hours of sunset, but different prey have different peak activity periods); spatial partitioning, such as between-, within-, and below-canopy foragers; and flight patterns, such as slow vs. fast flying, maneuverability, and hovering.

Although Townsend's big-eared bat has been characterized as a "relatively late flyer" by Kunz and Martin (1982), there are numerous observations that individuals leave roosts promptly at dusk like other species (Szewczak, pers. comm. 2012). Further, there is no information

[1] Both *M. coliolabrum* and *M. melanorhinus* were once considered subspecies of *M. leibii*, which is the species listed in Kunz and Martin (1982), but Wilson and Reeder (2005) list both as distinct species.

to suggest resource partitioning or direct competition for prey with other species. Although, artificial lighting may affect competitive predator-prey relationships among some bats (e.g., Frank 1988; Longcore and Rich 2004), the potential for this occurring in Townsend's big-eared bats is low because this species roosts and forages away from human-developed areas (Szewczak, pers. comm. 2012).

Population Status and Trends

Global: Apparently secure (NatureServe 2011)
State: Vulnerable to imperiled (CDFG 2011)
Within Plan Area: Same as state

Townsend's big-eared bat is a California Species of Special Concern, but there are little systematic data to quantitatively assess population status and trends (e.g., numbers of individuals). However, past studies have shown a broad-ranging decline in the species through large parts of its range in the western United States (i.e., mainly the *C .t. townsendii* and *C. t. pallescens* subspecies). Human disturbance has eliminated most historical roosting sites in California and all known previously occupied limestone caves in the state have been abandoned (see discussion in Threats and Stressors). The census by Pierson and Rainey (CDFG 1998) in California, conducted from 1987 to 1991, found substantial population declines over the previous 40 years, with a 52% loss in the number of maternity colonies, a 44% decline in the number of available roosts, a 55% decline in the total number of animals (primarily adult females), and a 32% decrease in the average size of remaining colonies. Fate of roosts sites was related to the type of roost, with 88% of roosts in buildings no longer available, and 50% of roosts in caves and 57% in mines no longer used. Pierson and Rainey (CDFG 1998) also reviewed population information for other western states as of 1998, summarized below.

- Arizona – 13 verified maternity roosts, representing 10 separate colonies, with a total population of about 1,000 adult females. Two cave populations extirpated and another declined by 50% in 2 years after its cave roost was commercialized. Another population historically supporting several hundred adult females numbered fewer than 100 individuals.

- Colorado – hibernaculum with more than 500 individuals in December 1968 apparently reduced to only a few animals. Only four maternity sites had been documented in Colorado since 1970, and the largest had only approximately 80 adult females.
- New Mexico – >10,000 individuals hibernating in a timber-lined 100-meter-deep mine shaft in 1992. The shaft was burned by vandals, and several hundred dead animals were seen still hanging from the walls, and thousands more were presumed dead.
- Idaho – surveys of known hibernating sites indicate a 60% population decline since 1987.
- Nevada – surveys conducted in the late 1980s to late 1990s in 96,000 km^2 of northeastern Nevada revealed only two small maternity sites.
- Oregon/Washington – severe population declines for both summer and winter populations in Oregon and Washington have been well documented. Known sites in Oregon and Washington contained approximately 2,700 and 800 adult females, respectively.

The isolated populations of *C. t. ingens* and *C. t. virginianus* are considered to be in danger of extinction because of their susceptibility to human disturbance (Kunz and Martin 1982), and both subspecies were federally listed as endangered in 1979 (44 FR 69206–69208).

Threats and Environmental Stressors

Townsend's big-eared bats are very sensitive to human disturbances, and a single disturbance of a maternity roost or hibernation site may cause abandonment (Zeiner et al. 1990; Kunz and Martin 1982). All known limestone cave sites in California, for example, have been abandoned (Zeiner et al. 1990). Sherwin et al. (2000) found that occupied day roosts were typically subject to little human disturbance. As discussed in Population Trends and Status, there has been a significant decline in occupied Townsend big-eared bat roosts in California. The primary cause for the observed declines was determined to be human disturbance of roosting sites (CDFG 1998). As of 1998, 37 known maternity colonies had a total population of

approximately 4,250 adult females, but only three of these colonies were considered adequately protected. Declines were also indicated at four important hibernacula for which past population data were available (CDFG 1998). The selection of relatively cold parts of caves near entrances and where there is good ventilation during hibernation makes Townsend's big-eared bats sensitive to human disturbance (including deliberate vandalism and extermination) during a period when they would be least likely to respond quickly. Also, they tend to hang from ceilings and walls in exposed parts of roosts, making them more susceptible to disturbance (CDFG 1998). It is important that hibernacula be protected from human disturbance because animals can be aroused from hibernation and forced to use fat stores necessary for hibernation.

Pierson and Rainey (CDFG 1998) provided specific information for threats to roosts in the Plan Area. The active roosts in mines on public lands in the eastern Sierra area were considered to be at risk from recreation, mine closure for hazards, and reactivation of old mining claims. An occupied mine at the China Lake Naval Air Weapons Station was vandalized in 1988 and has not been since reoccupied. Other mines have shown evidence of extensive recreational use. Even the colony at Death Valley National Monument was vandalized in 1993, greatly reducing the number of individuals using the site. In the Providence Mountains, the Mitchell Caverns colony located in the State Park was excluded from using the site in 1970 when a bat-proof gate was installed, but replacement of the gate in 1993 resulted in rapid reoccupation. Reactivation of mining in Macedonia Canyon has excluded the species, but individuals appeared to relocate to another mine. In the Colorado River Basin and eastern Mojave Desert, Townsend's big-eared bat was once common at many mine sites, and three maternity sites were known, including the Alice Mine with the largest known colony (>1,000 individuals) in California. Surveys in 1990 and 1992 found only one small maternity site in 1990 but none in 1992. Abandoned mines in this region are subject to intensive recreation, but other apparently undisturbed mines also were unoccupied. Pierson and Rainey (CDFG 1998) suggest the agricultural conversion has reduced foraging habitat and that pesticides may be affecting this species in the region.

Several recent studies have documented substantial mortality of bats at wind facilities (e.g., Baerwald and Barclay 2009; Cryan 2011; Cryan and Barclay 2009). Despite fairly extensive monitoring, with many documented fatalities of other bat species (primarily migrant species), as of 2004, no Ozark or Virginia big-eared bats had been known to be killed at wind facilities (or at communications towers) (Johnson and Strickland 2004). In 2010, TetraTech also reported no documented fatalities of Townsend's big-eared bats at wind facilities (TetraTech EC Inc. 2010). A general review of the wind facility–related literature also failed to reveal evidence for, or discussions of, Townsend's big-eared bat fatalities or assessed risks at wind facilities (e.g., Baerwald and Barclay 2009; Cryan 2011; Cryan and Barclay 2009; Cryan and Brown 2007; Johnson and Strickland 2004; Johnson and Erickson 2008; Kuvlesky et al. 2007; Piorkowski and O'Connell 2010). Nonetheless, the U.S. Fish and Wildlife Service (USFWS) has expressed concern about the potential for fatalities of the endangered Virginia big-eared bats from wind facilities in the eastern United States as they move between caves (e.g., see Johnson and Strickland 2004). Big-eared bats in the Plan Area similarly could be at elevated risk of turbine strikes or other associated causes (e.g., barotrauma) if a wind facility were located within a few miles of a day roost site (where most foraging activity occurs), and strikes would most likely occur during emergence, return to the day roost, or when seeking a night roost between bouts of foraging. Risk of strikes may also be higher when bats are moving between maternity roosts and hibernacula in the fall and spring and when young are dispersing from the maternity roost in late summer.

Conservation and Management Activities

Townsend's big-eared bat is addressed in the West Mojave Plan (BLM 2005). Under Alternative A (the Proposed Action – Habitat Conservation Plan), BLM would implement several conservation measures for Townsend's big-eared bat and other bat species, including:

- Protection of all significant roosts (defined as maternity and hibernation roosts supporting 10 or more individuals) by installing gates over mine entrances and restricting human access. The West Mojave Plan identified two significant maternity roosts and two significant hibernation roosts for Townsend's big-eared bat on BLM-managed lands.

- Protection of bat roosts in the Pinto Mountains by gating known and new significant roosts and notifying claim holders on BLM lands containing significant roosts.
- Continued fencing around (but not over) open, abandoned mine features to provide bats access to roosts and to reduce hazards to the public.
- Required surveys for bats by applicants seeking discretionary permits for projects that would disturb natural caves, cliff faces, mine features, and abandoned buildings or bridges to determine whether significant roost sites are present.
- Safe eviction of bats at a non-significant roost (i.e., less than 10 individuals) prior to disturbance or removal.

BLM would also conduct monitoring and adaptive management for Townsend's big-eared bat. Monitoring actions include:

- Determining bat numbers in all significant roosts
- Conducting periodic surveys in the northern part of the planning area with high potential for containing significant roosts
- Determining and reporting the effectiveness of mitigation measures providing for safe exit of bats
- Reporting take from approved projects that impact bats under to California Department of Fish and Game (CDFG) and USFWS
- Monitoring population numbers using bat houses if installed.[2]

Adaptive management measures include:

- Gating mines where new significant roosts are found
- Installing bat houses in locations, where appropriate, if populations decline or are threatened[3]
- Case-by-case review of newly detected significant roosts near open routes within riparian and desert wash habitat. Corrective actions would be taken within the foraging habitat if

[2,3] The independent scientific reviewer for this profile (J. Szewczak, pers. comm. 2012) indicates that bat houses would not typically provide suitable habitat for Townsend's big-eared bat because this species requires space, not cervices. An artificial roost would have to be a cave-like structure or a building-size roost.

the new roosts are impacted by open routes or new routes would be established to avoid the habitat.

In addition, as a BLM sensitive species, Townsend's big-eared bat is addressed under other land use actions undertaken by BLM. In accordance with the BLM's "6840 – Special Status Species Management" manual, the objectives for sensitive species policy are:

> To initiate proactive conservation measures that reduce or eliminate threats to Bureau sensitive species to minimize the likelihood of and need for listing of these species under the ESA (BLM 2008).

Under this policy BLM must consider the impact of actions on sensitive species, including outcomes of actions (e.g., land use plans, permits), strategies, restoration opportunities, use restrictions, and management actions necessary to conserve BLM sensitive species.

Townsend's big-eared bat is also addressed in the Military Integrated Natural Resources Management Plans (INRMP) for the China Lake Naval Air Weapons Station (NAWS and BLM 2004) and the Marine Air Ground Task Force Training Command Marine Corps Air Ground Combat Center, Twentynine Palms (MAGTFTC MCAGCC 2007). As a designated sensitive species in these INRMPs, Townsend's big-eared bat is provided protection and management considerations during the land use planning process defined in the China Lake Comprehensive Land Use Management Plan and military training operations at Twentynine Palms. If it is determined to be at risk from a proposed project or training activities, efforts are made to avoid and minimize impacts. For example, at Twentynine Palms, four bat gates have been installed in three mines to allow bats access to roosts without disturbance from humans. The Twentynine Palms INRMP also includes three objectives:

- Monitoring current bat gates to inspect for trespass and condition
- Evaluating mine entrances for installation of bat gates to those mines that are exceptional bat habitat but not culturally significant
- Evaluating modification of bighorn sheep guzzlers for use by bats and other wildlife to enhance habitat value.

Data Characterization

Although Pierson and Rainey (CDFG 1998) conducted a thorough review of roosting sites for Townsend's big-eared bat, this information is dated. Also, in the Plan Area the current distribution and status of roosts is not well understood. For example, Townsend's big-eared bats may be using deep mine shafts that have not been accessed by qualified biologists (CDFG 1998) or monitored for bats entering or leaving (Szewczak, pers. comm. 2012).

Management and Monitoring Considerations

The primary management and monitoring consideration for Townsend's big-eared bat is protection of day and night roosts from disturbance that may cause abandonment. This species is very sensitive to human disturbance because it tends to roost at the entrances of caves and may be found hanging from ceilings and walls were it is susceptible to disturbance. Occupied maternity and winter roosts should be considered a highly valuable resource, and impacts should be avoided. Maintaining these sites requires protecting them from human disturbances and adjacent land uses that could cause direct mortality or injury of big-eared bats or abandonment of the roost site. Protection of riparian habitats and desert wash near roost sites (e.g., within 5 miles) is also important because these areas are important prey resource areas.

Another consideration for Townsend's big-eared bat for monitoring and management is that their echolocation signals are relatively weak. (Their large pinnae amplify weak echoes from their low amplitude calls, which enable them to more closely approach their primary prey of moths, many of which can hear, and defensively react, to bat echolocation calls [Szewczak, pers. comm. 2012]). O'Farrell and Gannon (1999) found that the big-eared bat was more effectively sampled using capture methods because their calls could only be detected at less than about 5 meters (16 feet) from the bat with the existing bat detectors. New generation acoustic detectors are more sensitive and can be deployed for long time periods, and therefore are better able to detect the species (Szewczak, pers. comm. 2012). Nonetheless, monitoring for this species may remain a challenge

because the probability of detection could still be limited without broad spatial coverage of monitoring stations due to its restricted area around the primary roost used for foraging (Szewczak, pers. comm. 2012). Further, this species is difficult to physically capture due to its slow flight and high maneuverability (Szewczak, pers. comm. 2012).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Townsend's big-eared bat, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 16,824,190 acres of modeled suitable habitat for Townsend's big-eared bat in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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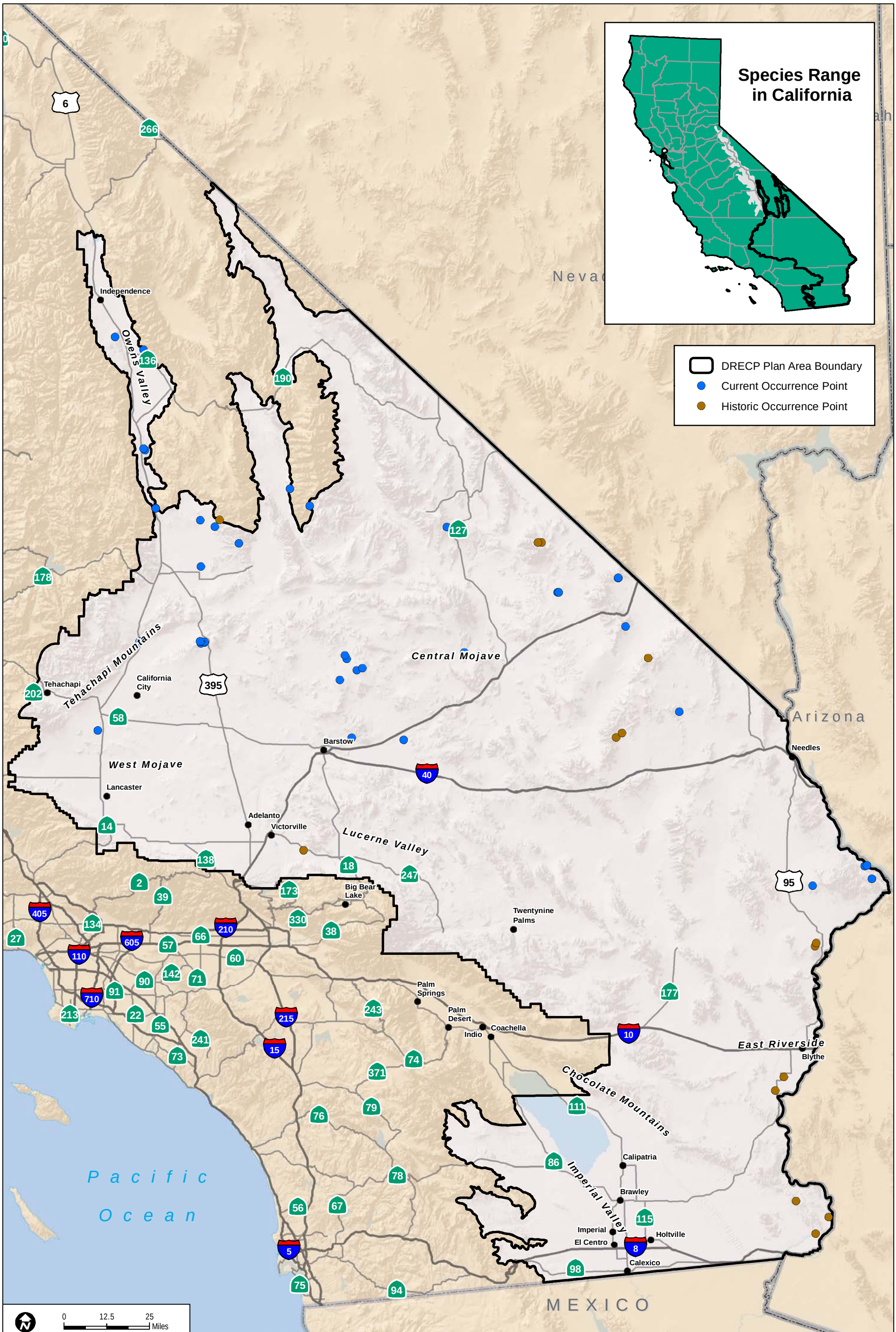

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-M07

Townsend's Big-eared Bat Occurrences in the Plan Area

Tricolored Blackbird (*Agelaius tricolor*)

Photo courtesy of Rob Schell Photography.

Legal Status

State: Species of Special Concern
Federal: Bureau of Land Management, U.S. Fish and Wildlife Service Bird of Conservation Concern
Critical Habitat: N/A
Recovery Planning: N/A
Notes: Previously listed as Category 2 Candidate Species in 1991 (56 FR 58804–58836).

Taxonomy

Tricolored blackbird (*Agelaius tricolor*) is endemic to the west coast of North America and primarily to California. No subspecies are currently recognized (Beedy and Hamilton 1999). Songs of male tricolored blackbirds are not regionally distinguishable, unlike those of some red-winged blackbird (*A. phoeniceus*) populations in California (Beedy and Hamilton 1999). Banding studies by Neff (1942, cited in Beedy and Hamilton 1999), DeHaven and Neff (1973, cited in Beedy and Hamilton 1999), and DeHaven et al. (1975a, cited in Beedy and Hamilton 1999) found no tricolored blackbirds from elsewhere among populations breeding from Santa Barbara County south to Baja California and east to the Sonoran Desert, suggesting potential for a separate metapopulation in southern California. Furthermore, more recent studies have found this species in Southern California are not genetically distinct (Pollinger and Berg, in preparation, cited in Feenstra 2012).

Distribution

General

Tricolored blackbird is largely endemic to California, and more than 90% of the population occurs in the state (Churchwell et al. 2005). Population surveys and banding studies of tricolored blackbird in the

Central Valley from 1969 through 1972 concluded that their geographic range and major breeding areas were unchanged since the mid-1930s (DeHaven et al. 1975b).

In any given year, more than 75% of the breeding population can be found in the Central Valley (Hamilton 2000), increasingly concentrated in the San Joaquin Valley. This trend appears to be continuing; the latest statewide survey found 88% of the 2011 breeding population concentrated in large colonies in Merced, Kern, and Tulare counties (Kyle and Kelsey 2011). Much smaller colonies are found in southern coastal counties and west of the desert in Southern California (Beedy and Hamilton 1999). The species also breeds in marshes of the Klamath Basin in Siskiyou and Modoc counties, and Honey Lake Basin in Lassen County. Small breeding populations also exist at scattered sites in Oregon, Washington, Nevada, and the western coast of Baja California (Beedy and Hamilton 1999) (Figure SP-B14). During winter, virtually the entire population of the species withdraws from Washington, Oregon (although a few remain), Nevada, and Baja California, and wintering populations shift extensively within their breeding range in California (Beedy and Hamilton 1999).

Distribution and Occurrences within the Plan Area

Historical

Tricolored blackbird historical breeding range in California included the Sacramento and San Joaquin valleys, lowlands of the Sierra Nevada south to Kern County, the coast region from Sonoma County to the border of Mexico, and sporadically on the Modoc Plateau (Dawson 1923; Neff 1937; Grinnell and Miller 1944).

Tricolored blackbird was described as locally common in the coastal area of Southern California and also bred on the western edge of the desert in Antelope Valley (Garrett and Dun 1981). Birds were resident year-round, dispersing only short distances from the breeding colonies (Garrett and Dun 1981).

There are four historical (i.e., pre-1990) occurrences recorded in the Plan Area and an additional four records with an unknown

observation date (CDFW 2013; Dudek 2013). These occurrences are located in the Harper Lake area, Palmdale/Lancaster area, and in the southwestern portion of Edwards Air Force Base (AFB) (Figure SP-B14).

Recent

[Note to Reader: additional verification on nature of occurrence data (colonies versus individuals) is ongoing as is the integration of recent Tricolored Blackbird Working Group data. This section will be updated as data become available.]

Tricolored blackbirds breed in lowland areas in the western and central portions of the Plan Area (Figure SP-B14). Breeding colonies occur in eastern Kern County from Ridgecrest along the base of the Tehachapi Mountains to Antelope Valley, around Palmdale and Lancaster in northeast Los Angeles County, and east of Barstow in San Bernardino County. There are 47 recent (i.e., since 1990) occurrences for the Plan Area (CDFW 2013; Dudek 2013). These occurrences generally are located in the Lancaster/Palmdale area, in the southwestern portion of Edwards AFB, just north of State Highway 138, along State Highway 158 in the Tehachapi Mountain foothills, west and south of Red Rock Canyon State Park, along the Trona Road cutoff north of State Highway 395, in the southern portion of the China Lake Naval Air Weapons Station north of Ridgecrest, and along the Mojave River east of Barstow (Figure SP-B14).

Natural History

Habitat Requirements

Breeding tricolored blackbirds form large colonies, typically in freshwater wetlands dominated by cattails (*Typha* spp.) or bulrushes (*Schoenoplectus* spp.) and thorny vegetation such as Himalayan blackberry (*Rubus armeniacus,* formerly *R. discolor*)(Churchwell et al. 2005). They may also nest in willows (*Salix* spp.), thistles (*Cirsium* and *Centaurea* spp.), and nettles (*Urtica* spp.) (Beedy and Hamilton 1999). They forage away from their breeding grounds in rice fields, lightly grazed pasture, dairies, or alfalfa fields. With the conversion of wetlands to arable land, tricolored blackbirds began exploiting the

rich agricultural fields created by the transition to farming. Recently, the species has been using dairies, which contain many of the necessary characteristics for breeding. As a result, the expanding dairy industry in the San Joaquin Valley has led to a shift in distribution and the concentration of species into mega-colonies of tens of thousands of birds. In 2008, 50% of breeding tricolors in California were observed nesting in silage fields (Kelsey 2008).

Tricolored blackbirds have three basic requirements for selecting their breeding colony sites: open, fresh water; a protected nesting site, provided by flooded, thorny, or spiny vegetation; and a suitable foraging space providing adequate insect prey within a few miles of the nesting colony (Hamilton et al. 1995; Beedy and Hamilton 1997, 1999; Churchwell et al. 2005). Almost 93% of the 252 breeding colonies reported by Neff (1937) were in freshwater marshes dominated by cattail and bulrush species. In contrast, only 53% of the colonies reported during the 1970s were in cattails and bulrushes (DeHaven et al. 1975a).

An increasing percentage of tricolored blackbird colonies in the 1980s and 1990s were reported in Himalayan blackberry (Cook 1996), and some of the largest recent colonies have been in silage and grain fields (Hamilton et al. 1995; Beedy and Hamilton 1997; Hamilton 2000). Other vegetation used by nesting tricolored blackbirds includes giant cane (*Arundo donax*), safflower (*Carthamus tinctorius*) (DeHaven et al. 1975a), tamarisk (*Tamarix* spp.), elderberry (*Sambucus* spp.), poison-oak (*Toxicodendron diversilobum*), and riparian scrub and forests (e.g., *Salix, Populus, Fraxinus*) (Beedy and Hamilton 1999).

Ideal foraging conditions for tricolored blackbird is created when shallow flood irrigation, mowing, or grazing keeps the vegetation at an optimal height (<15 cm [<5.9 inches]) (Tricolored Blackbird Working Group 2007). Preferred foraging habitats include agricultural crops such as rice, alfalfa, irrigated pastures, and ripening or cut grain fields (e.g., oats, wheat, silage, and rice), as well as annual grasslands, cattle feedlots, and dairies. Tricolored blackbird also forages in remnant native habitats, including wet and dry vernal pools and other seasonal wetlands, riparian scrub habitats, and open marsh borders (Tricolored Blackbird Working Group 2007). See Table 1 for a summary of tricolored blackbird habitat associations.

Table 1. Habitat Associations for Tricolored Blackbird

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Wetland	Breeding	Primary	Cattails, bulrushes, willows, Himalayan blackberries (recent shift), thistles, nettles, and other spiny or thorny plants	Beedy and Hamilton 1999
Riparian	Breeding	Primary	Riparian woodland and scrub	Beedy and Hamilton 1999
Agricultural	Foraging	Secondary	Open pastures, silage, grain fields, mowed alfalfa, pastures, dairies	Beedy and Hamilton 1999
Herbaceous dominated	Foraging	Secondary	Native and non-native annual grasslands	Beedy and Hamilton 1999

Foraging Requirements

Tricolored blackbirds forage primarily in artificial habitat with ideal foraging conditions created in shallow flooded fields. Preferred foraging habitat includes crops, annual grasslands, cattle feedlots, and dairies (Beedy and Hamilton 1999). Foods delivered to tricolored blackbird nestlings include beetles and weevils, grasshoppers, caddisfly larvae, moth and butterfly larvae, and dragonfly larvae (Orians 1961a; Crase and DeHaven 1977; Skorupa et al. 1980; Beedy and Hamilton 1999). Breeding-season foraging studies in Merced County showed that animal matter makes up about 91% of the food volume of nestlings and fledglings, 56% of the food volume of adult females, and 28% of the food volume of adult males (Skorupa et al. 1980).

Adults may continue to consume plant foods throughout the nesting cycle, but they also forage on insects and other animal foods. Immediately before and during nesting, adult tricolored blackbirds are often attracted to the vicinity of dairies, where they take high-energy items from livestock feed. Adults with access to livestock feed (such as cracked corn) begin providing it to nestlings when they are

about 10 days old (Hamilton et al. 1995). More than 88% of all winter food in the Sacramento Valley is plant material, primarily seeds of rice and other grains, but also weed seeds (Crase and DeHaven 1978). In winter, tricolored blackbird often associates with other blackbird species (*Agelaius* spp.; *Euphagus* spp.), but flocks as large as 15,000 individuals (almost all tricolored blackbirds) may congregate at one location and disperse to foraging sites (Beedy and Hamilton 1999).

Reproduction

Tricolored blackbird is closely related to red-winged blackbird, but the two species differ substantially in their breeding ecology. Red-winged blackbird pairs defend individual territories, while tricolored blackbirds are among the most colonial of North American passerine birds (Bent 1958; Orians 1961a, 1961b, 1980; Orians and Collier 1963; Payne 1969; Beedy and Hamilton 1999). As many as 20,000 or 30,000 tricolored blackbird nests have been recorded in cattail marshes of 4 hectares (9 acres) or less (Neff 1937; DeHaven et al. 1975a), and individual nests may be built less than 0.5 meter (1.5 feet) apart (Neff 1937). The tricolored blackbird colonial breeding system may have adapted to exploit a rapidly changing environment where the locations of secure nesting habitat and rich insect food supplies were ephemeral and likely to change each year (Orians 1961a; Orians and Collier 1963; Collier 1968; Payne 1969). See Table 2 for a summary of seasonal migration, colony formation, and breeding.

Table 2. Key Seasonal Periods for Tricolored Blackbird

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Colony Formation			✓	✓	✓							
Breeding			✓	✓	✓	✓	✓					
Migration			✓	✓	✓	✓	✓	✓	✓	✓		
Other							✓	✓	✓	✓		

Source: Beedy and Hamilton 1999.

Spatial Activity

During the breeding season, tricolored blackbird exhibits itinerant breeding, commonly moving to different breeding sites each season (Hamilton 1998). In the northern Central Valley and northeastern California, individuals move after their first nesting attempts, whether successful or unsuccessful (Beedy and Hamilton 1997). Banding studies indicate that significant movement into the Sacramento Valley occurs during the post-breeding period (DeHaven et al. 1975b).

During winter, virtually the entire population withdraws from Washington, Oregon (although a few remain), Nevada, and Baja California, and wintering populations shift extensively within their breeding range in California (Beedy and Hamilton 1999). Tricolored blackbird numbers decrease in the Sacramento Valley and increase in the Sacramento–San Joaquin River Delta and northern San Joaquin Valley (Neff 1937; Orians 1961a; Payne 1969; DeHaven et al. 1975b). By late October, large flocks of tricolored blackbird also congregate in pasturelands in southern Solano County and near dairies on Point Reyes Peninsula in Marin County (Beedy and Hamilton 1999). Other birds winter in the central and southern San Joaquin Valley. Concentrations of more than 15,000 wintering tricolored blackbirds may gather at one location and disperse up to 32 kilometers (20 miles) to forage (Neff 1937; Beedy and Hamilton 1999). Individual birds may leave winter roost sites after fewer than 3 weeks and move to other locations (Collier 1968), suggesting winter turnover and mobility. In early March and April, most birds vacate wintering areas in the Central Valley and along the coast, and move to breeding locations in the Sacramento and San Joaquin valleys (see Table 3) (DeHaven et al. 1975b). In the Plan Area, tricolored blackbirds appear to be more sedentary and winter close to their breeding colonies (Garret and Dunn 1981).

Table 3. Movement Distances for Tricolored Blackbird

Type	Distance/Area	Location of Study	Citation
Male territory (within colony)	20 to 35 square feet (0.8 to 3.25 m^2)	California	Lack and Emlen 1939; Orians 1961a

Type	Distance/Area	Location of Study	Citation
Dispersal	33% recovered within 10 miles of natal colonies	California	DeHaven et al. 1975b
Home range	May range widely in flocks to over 9 miles from active colony	California	Beedy and Hamilton 1999

Ecological Relationships

Tricolored blackbird occupies a unique niche in the Central Valley/coastal marshland ecosystems. In areas where the number of tricolored blackbirds is high, they are both aggressively and passively dominant to—and often displace—sympatric marsh nesting species, including red-winged and yellow-headed blackbird (*Xanthocephalus xanthocephalus*) (Orians and Collier 1963; Payne 1969).

Nest predation is a major cause of nesting failure at some tricolored blackbird colonies. Historical accounts documented the destruction of nesting colonies by a diversity of avian, mammalian, and reptilian predators. Recently, especially in permanent freshwater marshes of the Central Valley, entire colonies (>50,000 nests) have been lost to black-crowned night-heron (*Nycticorax nycticorax*), common raven (*Corvus corax*), coyote (*Canis latrans*), and other predators (Beedy and Hayworth 1992; Beedy and Hamilton 1999).

Population Status and Trends

Global: Declining (Beedy and Hamilton 1997, 1999)
State: Declining (Beedy and Hamilton 1997, 1999)
Within Plan Area: Unknown

The U.S. Fish and Wildlife Service (USFWS), the California Department of Fish and Game (CDFG), and California Audubon cosponsored intensive tricolored blackbird surveys (carried out by volunteers in suitable habitats throughout California) in 1994, 1997, 1999, 2000,

2004, 2008, and 2011 (Hamilton et al. 1995; Beedy and Hamilton 1997; Hamilton 2000; Green and Edson 2004; Churchwell et al. 2005; Kyle and Kelsey 2011). Local, regional, and statewide tricolored blackbird populations experienced major declines between 1994 and 2004. Statewide totals of adults in four late-April surveys covering all recently known colony sites were 369,359 (1994); 237,928 (1997); 104,786 (1999); 162,508 (2000); and >130,000 (low estimate for 2004). Several areas that historically supported large (>2,000 individuals) colonies in the Central Valley no longer have birds present (Green and Edson 2004; Hamilton 2004).

The Audubon species account for tricolored blackbird also reports a decline from 1994 to 2000, with numbers stabilizing since that time (Audubon 2012). However, results of the Audubon California 2011 statewide survey (Kyle and Kelsey 2011) show a dramatic drop in the species population numbers throughout the state: in all, slightly fewer than 260,000 birds were observed compared to 395,000 in the 2008 survey, a 33% decrease in the population.

Threats and Environmental Stressors

The greatest threats to this species are the loss and degradation of habitat as a result of human activities (Beedy and Hamilton 1999). One of the main causes for population decline has been the near elimination of native cattail wetland complexes throughout central California by agricultural expansion and conversion of wetlands (Kyle and Kelsey 2011). Tricolored blackbird subsequently exploited the croplands that replaced their native habitat. Because of the increasing importance of agricultural fields to the species and the use of Triticale (a hybrid of wheat and rye grown as silage on dairies) as nesting habitat, tricolored blackbirds are at high risk when farmers need to cut their silage in the middle of the tricolored blackbird breeding effort. Entire colonies of up to tens of thousands of nests have been destroyed by harvesting and plowing of agricultural lands (Beedy and Hamilton 1999).

In addition to direct loss and alteration of habitat, other factors also threaten tricolored blackbird populations (Beedy and Hamilton 1999). These factors include predation of fledglings and adults by black-crowned night herons and ravens (Hamilton 2004). In addition,

the application of herbicides and pesticides may affect the nesting success of colonies in agricultural areas (Beedy and Hamilton 1999). Various poisons and contaminants have caused mass mortality, including poisoning by strychnine, selenium, and spraying with mosquito abatement oil (Beedy and Hayworth 1992; Beedy and Hamilton 1999; Beedy 2008).

Conservation and Management Activities

A variety of proposed and ongoing conservation and management activities are relevant to the tricolored blackbird in or near the Plan Area. The Western Riverside Habitat Conservation Plan/Natural Community Conservation Plan (HCP/NCCP), adjacent to the Plan Area, conserves 420 acres of suitable primary habitat and 66,510 acres of suitable secondary habitat.

The Tricolored Blackbird Working Group lists eight goals for the species, including habitat conservation and the protection of silage-nesting tricolored blackbirds (Tricolored Blackbird Working Group 2007). Protection of historical colonies should be prioritized and habitat managed to enhance the three habitat requirements described previously to encourage nesting. Churchwell et al. (2005) recommend water management and cited the success of the water bank Conservation Reserve Program, a voluntary program for agricultural landowners that promotes water storage until mid-July.

In 1993 and 1994, CDFG and USFWS purchased portions of crops to preserve several large colonies in Kings, Fresno, and Tulare counties. These and other actions are thought to have resulted in an additional 37,000 and 44,000 first-year adults to the 1994 and 1995 breeding seasons, respectively (as cited in Beedy and Hamilton 1999). USFWS may also provide compensation for delayed harvest to allow nestlings to fledge.

Preservation of wetlands and acquisition of agricultural lands for wetland restoration do not always benefit tricolored blackbirds because they are typically managed for waterfowl and other species in ways that do not provide suitable habitat for tricolored blackbirds, particularly in the nesting season.

The Tricolored Blackbird Working Group has set a long-term target of increasing the population to 750,000 birds, which will require the creation of new breeding habitat and the enhancement of existing colony sites on public and private lands (Kyle and Kelsey 2011).

Data Characterization

Statewide tricolored blackbird surveys were conducted in California in 1994, 1997, 1999, 2000, 2004, 2008, and 2011 (Hamilton et al. 1995; Beedy and Hamilton 1997; Hamilton 2000; Green and Edson 2004; Kyle and Kelsey 2011). Additional surveys include data on local distribution and population trends (Neff 1937; DeHaven et al. 1975a).

A relatively large amount of literature is available for the tricolored blackbird because it is a highly visible, colonial bird species of conservation concern, commonly associated with wetland habitat. Beedy and Hamilton (1999) provide a comprehensive review of information available on general natural history, behavior, distribution and population changes, known demographics and population regulation, and conservation and management. A range-wide management plan was developed in 1997 (Beedy and Hamilton 1997) and the Tricolored Blackbird Working Group released a conservation plan for tricolored blackbirds in 2007.

Management and Monitoring Considerations

A conservation plan for tricolored blackbirds was developed in 2007 by the Tricolored Blackbird Working Group. In addition to the conservation activities described above, the conservation plan outlines several management and monitoring priorities:

- Document the annual breeding, foraging, and wintering distribution and long-term population trends of the species
- Monitor reproductive success and adult survivorship to more effectively assess population viability
- Develop a strategic monitoring program using standardized methods that can be compared across time and geography, and adaptively changed for maximum effectiveness

- Identify environmental characteristics associated with breeding success
- Improve understanding of population dynamics and add to existing scientific understanding of the species
- Support and facilitate management-oriented research on public and private land.

To document seasonal and spatial movements, including site fidelity, several thousand tricolored blackbirds have been color banded, and observers are encouraged to submit sightings of banded birds.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for tricolored blackbird, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 277,915 acres of modeled suitable habitat for tricolored blackbird in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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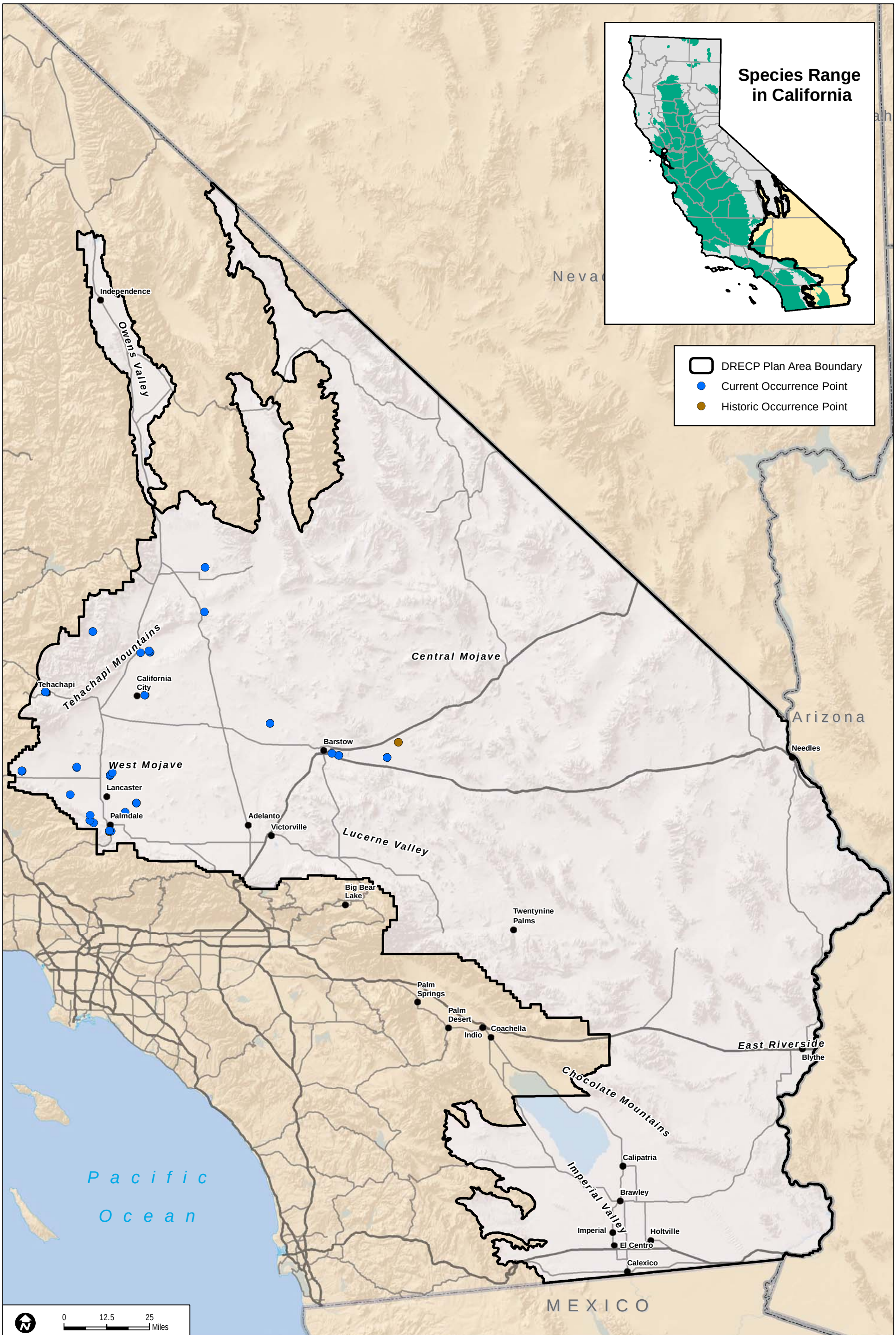

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B12

Tricolored Blackbird Occurrences in the Plan Area

Alkali Mariposa-Lily (*Calochortus striatus*)

Photo courtesy of Dr. Heath McAllister.

Legal Status

State: S2[1]
California Rare Plant Rank: 1B.2[2]
Federal: Bureau of Land Management Sensitive; U.S. Forest Service Sensitive
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

Alkali mariposa-lily (*Calochortus striatus*) is a perennial bulbiferous herb in the lily family (Liliaceae) (Jepson Flora Project 2011). Alkali mariposa-lily was described by S.B. Parish in 1902 (IPNI 2011). Although it appears that alkali mariposa-lily has been uniformly accepted as distinct since 1940, it was once considered by some to be synonymous with *C. palmeri* based partly on confusion of type specimens (Greene and Sanders 2006).

Alkali mariposa-lily stands approximately 1 to 4.5 decimeters (3.9 to 17.7 inches) in height (Munz and Keck 1968). A full physical description of the species can be found in the *Jepson eFlora* (Jepson Flora Project 2011) and Greene and Sanders (2006).

Distribution

General

Alkali mariposa-lily occurs in Southern California and western Nevada (Jepson Flora Project 2011). Within Southern California, alkali mariposa-lily occurs in Tulare, Kern, Los Angeles, and San Bernardino counties (CNPS 2011). More specifically, this species

[1] **S2:** Imperiled.
[2] **1B:** Rare, threatened, or endangered in California and elsewhere; **X.2:** Fairly threatened in California.

occurs in southern Sierra Nevada; in the Mojave Desert; at the north base of the San Bernardino and San Gabriel Mountains; and in the southern San Joaquin Valley (Figure SP-P01; CDFW 2013a; Jepson Flora Project 2011; Munz and Keck 1968). Of the 102 total occurrences recorded in the California Natural Diversity Database (CNDDB), 87 are in the Plan Area (CDFW 2013a). It is rare in Nevada, with only three occurrences recorded (NNHP 2001).

Distribution and Occurrences within the Plan Area

Historical

Of the 294 localities documented in the Plan Area, 18 are considered historical. Localities considered historical have not been observed since 1989, or were recorded in 2005, but have been extirpated or possibly extirpated. They range from Kelso Valley southeast to Twentynine Palms with most localities at or near Edwards Air Force Base (AFB) (Figure SP-P01) (CDFW 2013a). The 276 remaining localities recorded since 1990 and presumed extant are discussed below.

Recent

The recent localities (i.e., since 1990) of alkali mariposa-lily reported in the Plan Area by the CNDDB range from Red Rock Canyon State Park southeast to Joshua Tree National Park. The majority of localities are located on or in the vicinity of Edwards AFB (CDFW 2013a). Alkali mariposa-lily populations are most concentrated in the metapopulation that ranges from Lancaster to Edwards AFB (CDFW 2013a). A total of 126 localities are located on Edwards AFB, and 120 of these are managed by the Department of Defense (DOD), while 6 are privately owned. Other public localities include two on lands managed by the Department of Parks and Recreation (DPR) at Red Rock Canyon State Park, four on lands managed by Los Angeles County, one on lands managed by the National Park Service (NPS) at Joshua Tree National Park, one on lands managed by the BLM, and 15 on lands managed by Rosamond Community Services. About 108 localities are on privately owned land and ownership is unknown for 19 localities (CDFW 2013a).

Natural History

Habitat Requirements

Alkali mariposa-lily grows in seasonally moist alkaline habitats such as alkaline meadows and seeps, and ephemeral washes, within chaparral, chenopod scrub, and Mojavean desert scrub (CNPS 2011; CDFW 2013a; Jepson Flora Project 2011). Alkali mariposa-lily grows in calcareous sandy soil (Fiedler 1985, cited in Greene and Sanders 2006). It prefers claypans and sand dunes, especially along drainages, in halophytic (associated with saline soils) saltbush scrub (Edwards AFB 2002). Periodic natural inundation is important to alkali mariposa-lily (Edwards AFB 2002), however, alkali mariposa-lily has been reported as absent from areas with surface salts or areas with permanent standing surface water (Mitchell 1988, cited in Greene and Sanders 2006). This species ranges in elevation from 224 to 5,240 feet (BLM 2010; CDFW 2013a).

Some associated species include saltgrass (*Distichlis spicata*), rushes (*Juncus* spp.), sedges (*Carex* spp.), beardgrass (*Polypogon* sp.), dock (*Rumex* sp.), alkali sacaton (*Sporobolus airoides*), beardless wildrye (*Elymus triticoides*), dwarf checkerbloom (*Sidalcea malviflora*), rabbitbrush (*Chrysothamnus* sp.), Baltic rush (*Juncus balticus*), and yellow sweetclover (*Melilotus indicus*) (CDFW 2013a). Table 1 lists primary habitat associations and parameters for the alkali mariposa-lily.

Table 1. Habitat Associations for Alkali Mariposa-Lily

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Chaparral, chenopod scrub, Mojavean desert scrub, meadows, and seeps	Primary	Calcareous sandy soils, alkaline, seasonally moist, 224 to 5,240 feet elevation	CNPS 2011; Greene and Sanders 2006; BLM 2010; CDFW 2013a

Reproduction

Alkali mariposa-lily blooms from April to June (CNPS 2011). Alkali mariposa-lilies have perfect flowers (i.e., which contain both the male

and female reproductive parts) (Tollefson 1992, cited in Greene and Sanders 2006). The plants arise from small membranous-coated bulbs. It is unknown whether reproduction is most commonly from seedling establishment or bulb division (Greene and Sanders 2006). Alkali mariposa-lily is pollinated by bees and flies (Tollefson 1992, cited in Greene and Sanders 2006). Although seed dispersal mechanisms for this species are unknown, seeds of some other species of *Calochortus* are gravity-dispersed (Miller et al. 2004).

Ecological Relationships

Other than the habitat associations and pollination by bees and flies described above, little is known of the life history and ecological relationships of alkali mariposa-lily.

Abundances of alkali mariposa-lily fluctuate substantially from year to year (NatureServe 2011). The bulb remains dormant and may not sprout in dry years, and the bulb may not compete well since the species is not found in stands of tall grasses (Greene and Sanders 2006).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed 2009)
State: S2, Imperiled (CDFW 2013b)

Abundance figures are complicated by large fluctuations from year to year, making population trends difficult to assess (NatureServe 2011). Despite its relatively wide distribution, the majority of the populations are small with the exception of the metapopulation that ranges from Lancaster to Edwards AFB (CDFW 2013a). A majority of the species' known occurrences are within California, with the exception of several occurrences in western Nevada.

At Red Rock Canyon in the Plan Area there were 44 plants reported in 1988, 13 in 1989, 133 in 1990, and 1,200 in 2003 (CDFW 2013a).

There are as many as 165,000 plants in 67 areas documented on Edwards AFB (Greene and Sanders 2006). Approximately 3,641 plants were observed in the center colony in 1995. Outside of Edwards AFB, approximately 400 plants were reported at three sites

around Lancaster in Los Angeles County in 1988, but this likely represents an underestimate of the population of alkali mariposa-lily in this area (Greene and Sanders 2006). In San Bernardino County, 50 to 100 plants were reported in 1982 at Box "S" Springs; fewer than 50 were reported at the edge of Cushenbury Springs in 1981; 30 to 40 plants were seen at Rabbit Springs in 1980; approximately 1,500 plants were reported in 1989 at Paradise Springs; and 2 plants were observed north of Paradise Springs in 1989 (CDFW 2013a). Also in San Bernardino County, fewer than 1,000 individuals were seen at Joshua Tree National Park in 2004 (CDFW 2013a).

Threats and Environmental Stressors

Alkali mariposa-lily is threatened by urbanization, grazing, trampling, road construction, hydrological alternations, and water diversions that lower the water table (CNPS 2011). It is also threatened by military operations, dumping, and grading (NatureServe 2011).

The greatest threat to alkali mariposa-lily is the lowering of water tables, which alters the seasonally moist alkaline habitat that this species requires. Urbanization in the Lancaster area is likely the second most severe threat to this species since the largest populations are concentrated near Lancaster (CDFW 2013a; Greene and Sanders 2006). Large populations along Sierra Highway that are primarily on private land and receive minimal protection are in danger of extirpation from expanding urbanization from Lancaster (CDFW 2013a; Greene and Sanders 2006).

Road construction also threatens this species. Historically, extirpations or population declines occurred with construction of Highway 18 at Whiskey Springs in the 1920s; with the expansion of Kaiser Cement, now Mitsubishi Cement Corp., in 1988 that included diking the flow of the spring and adding a parking lot at Cushenbury Springs; and with the development of a site with 300 plants near Radio Tower Meadow in 1989 (Greene and Sanders 2006; Deacon 2007).

Trampling and grazing may also severely reduce alkali mariposa-lily's reproductive capacity. A survey around Lake Isabella found that plants in ungrazed areas were taller, more robust, and more numerous than those in cattle grazed areas. From 1984 to 1991 low-

intensity horse grazing was tested at The Nature Conservancy's Kern River Preserve to determine the effect that soil disturbance and reduction of competing grasses and weeds would have on alkali mariposa-lily productivity. The grazed alkali mariposa-lily population did not experience a substantial increase or decrease compared to non-grazed control populations under low-intensity grazing (Tollefson 1992, cited in Greene and Sanders 2006). Pavlik et al. (2011) also documented strong impacts by mammalian herbivores on alkali mariposa-lily growth and reproduction in two consecutive years at Ash Meadows National Wildlife Refuge.

Although it may not be a more widespread problem, ongoing monitoring at The Nature Conservancy's Kern River Preserve suggests that competition from taller grasses, such as beardless wildrye (*Elymus triticoides*) and non-native barley (*Hordeum* spp.), may contribute to population declines (Tollefson 1992, cited in Greene and Sanders 2006).

Conservation and Management Activities

Thirty-nine alkali mariposa-lily occurrences are recorded on the Edwards AFB (CDFW 2013a). The Edwards Air Force Base Integrated Natural Resources Management Plan offers general conservation measures based on an ecosystem approach with a general goal of conserving and improving the habitat that would benefit all native species (Edwards AFB 2002). One of the goals included in the Plan is to review project plans to ensure drainage patterns are not changed in areas where listed or sensitive species, such as alkali mariposa-lily, occur (Edwards AFB 2002). Populations at the Nature Conservancy's Kern River Preserve populations are currently protected from development (Greene and Sanders 2006). Additional populations are on public and private lands with unknown conservation and management activities.

Data Characterization

Population trends are difficult to assess due to the large year-to-year fluctuations (NatureServe 2011). Some key components of the life history of the species have not been characterized. The most common mode of reproduction is not known. In addition, seed dispersal mechanisms are not known. However, because there is information

available for other similar species of *Calochortus*, and because there is recent occurrence information available for this species, there is sufficient information available to characterize this species.

Management and Monitoring Considerations

Because population numbers fluctuate widely year to year, alkali mariposa-lily requires long-term monitoring to detect population trends. Possible measures to maintain or restore the water table at its historic level and to remove or modify existing obstructions to natural spring or seep flows would benefit the species and should be discussed with land managers. Trampling and grazing by cows should be prevented by fencing known population sites. Although it has yet to be tested for this species, control of introduced weeds could reduce competition for resources, and thus improve reproductive capability (Greene and Sanders 2006). Protection from herbivors is essential for achieving stable or increasing population trends (Moore, pers. comm. 2012).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for alkali mariposa-lily, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 188,549 acres of modeled suitable habitat for alkali mariposa-lily in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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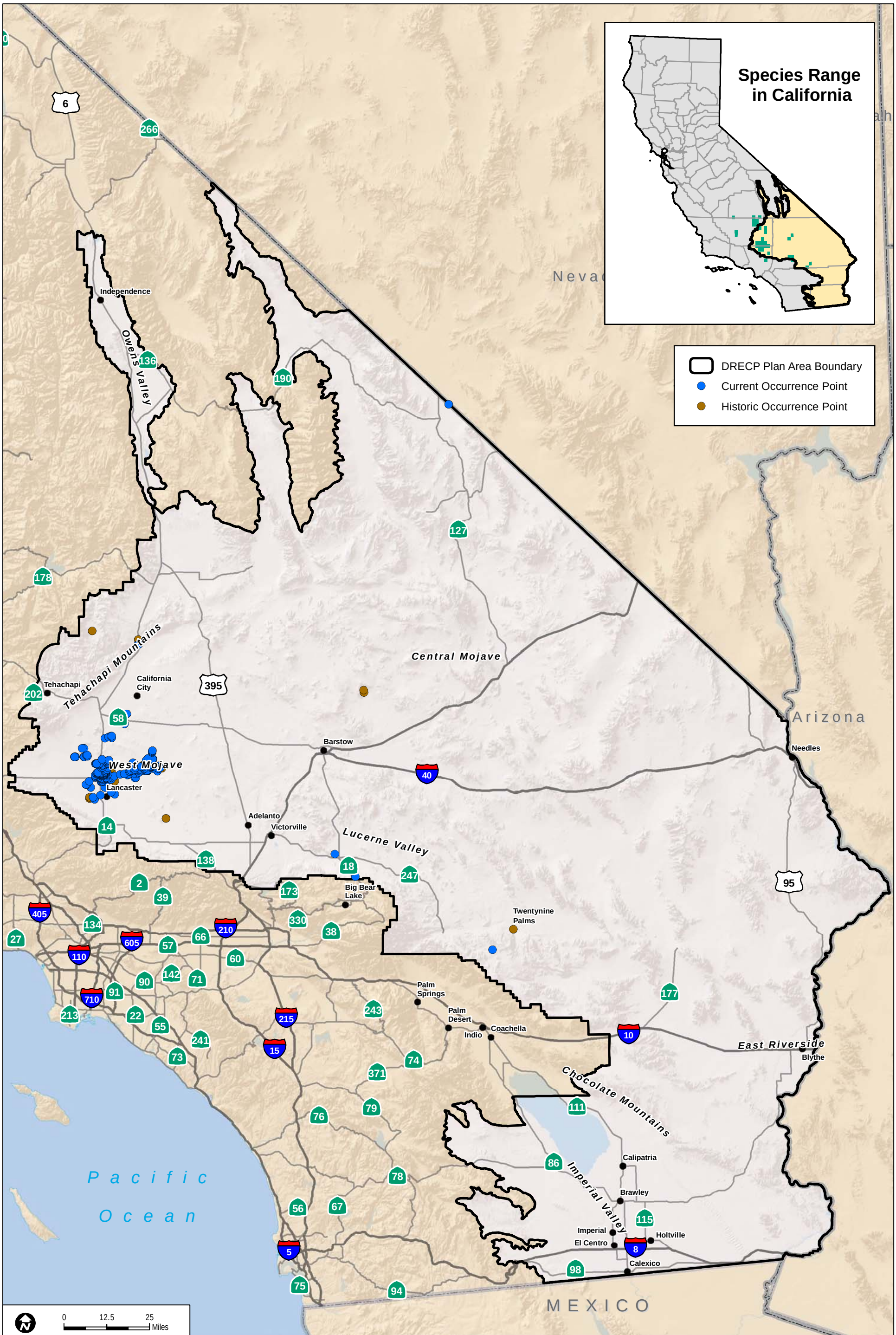

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P01

Alkali Mariposa Lily Occurrences in the Plan Area

Bakersfield Cactus (*Opuntia basilaris* var. *treleasei*)

Photo courtesy of Neal Kramer.

Legal Status

State: Endangered, S2.1[1]
California Rare Plant Rank: 1B.1[2]
Federal: Endangered, U.S. Forest Service Sensitive
Critical Habitat: N/A
Recovery Planning: Recovery Plan for Upland Species of the San Joaquin Valley, California (USFWS 1998)

Taxonomy

Bakersfield cactus (*Opuntia basilaris* var. *treleasei*) is a perennial stem succulent in the cactus family (Cactaceae) (Jepson Flora Project 2011; CNPS 2011). Bakersfield cactus was originally published as *Opuntia treleasei* by J.M. Coulter in 1896 (IPNI 2011). Bakersfield cactus was listed as *Opuntia treleasei* in the Federal Register notice announcing the endangered status of the species (55 FR 29361–29370). Bakersfield cactus has been consistently treated as a variety of *Opuntia basilaris* in every major California flora, including Munz and Keck (1959), Munz (1974), Hickman (1993), FNA(1993), and Baldwin et al. (2012), is to treat Bakersfield cactus as a variety of *O. basilaris*. since the publication of Jepson's 1936 *A Flora of California*.

Bakersfield cactus is low growing with stem segments approximately 9 to 20 centimeters (3.5 to 7.9 inches) long (USFWS 2011; Jepson Flora Project 2011). A full physical description of the species can be found in the Jepson eFlora (Jepson Flora Project 2011).

1 **S2:** Imperiled; **X.1:** Very threatened.
2 **1B:** Rare, threatened, or endangered in California and elsewhere; **X.1:** Seriously threatened in California.

Distribution

General

Bakersfield cactus occurs in the Tehachapi Mountain area and the southeastern San Joaquin Valley in Kern County, California (Figure SP-P02; Jepson Flora Project 2011). The historical distribution of Bakersfield cactus was likely more or less continuous east of Bakersfield, from Granite Station south to Comanche Point, east to Caliente, and west to Oildale (USFWS 1998, 2011). However, it is currently restricted to a limited area of central Kern County near Bakersfield in the southern San Joaquin Valley (USFWS 2011), and in the vicinity of Oak Creek and Mojave (Kentner, pers. comm. 2012). Approximately one-third of the historical population has been extirpated (USFWS 1998). The California Natural Diversity Database (CNDDB) includes 46 occurrences, of which 6 are in the Plan Area at 9 different localities (CDFW 2013a). However, there are a large number of records from the Plan area that were submitted to CNDDB in 2011, but have not been made publically available yet (Kentner, pers. comm. 2012; CDFW 2013a).

Following the recent discovery of the plants near Oak Creek, surveys for Bakersfield cactus were conducted on several thousand acres of proposed wind energy developments in the adjacent foothills of the eastern Tehachapi Mountains and the creosote brush and Joshua Tree woodlands of the desert areas to the east (Kentner, pers. comm. 2012).

Most of the individuals of the cactus population in this area are unambiguously identified as *Opuntia basilaris* var. *basilaris,* or beavertail cactus. However, the population is highly polymorphic and about a third of the individual plants display a varying number of morphological features that are characteristic of Bakersfield cactus (Kentner, pers. comm. 2012).

In 2010 and 2011, botanical surveys for proposed wind energy developments in the Tehachapi pass/Oak Creek area detected thousands of individual plants that were identified as Bakersfield cactus. The identification criteria were based on the recommendations of CDFG (Cypher 2011) which state that any plant with any one of several diagnostic characteristics of Bakersfield cactus should be considered to be the listed variety. Based on their

identification recommendations, CDFG has been requiring Incidental Take Permits and mitigation for the take of large numbers cactus in the vicinity of Oak Creek and Mojave within the Plan area (Kentner, pers. comm. 2012).

Point data for 1,244 individuals identified as Bakersfield cactus were submitted to CNDDB in the summer of 2011, and surveys have been ongoing since then. However, the CDFG identification criteria are controversial, and many of the identified plants appear to be intermediate between the varieties (Kentner, pers. comm. 2012).

Distribution and Occurrences within the Plan Area

Historical

Of the nine localities documented in the CNDDB within the Plan Area, one is considered historical with plants that have not been observed since 1934. This locality is mapped approximately 1 mile south of Fram (CDFW 2013a; Figure SP-P02). The historical locality in the Plan Area is east of the recent occurrences described below.

Recent

The eight recent localities of Bakersfield cactus reported in the Plan Area by the CNDDB occur at Oak Creek Pass in the Tehachapi Mountains, and near West Antelope Station and east of Bean Canyon at the foothills of the Tehachapi Mountains (Figure SP-P02; CDFW 2013a). Three of these localities are located on private land; ownership of the others is unknown (CDFW 2013a). Most of these localities are all very new, found in 2009 and 2010, and extend the variety's known range southeast since they occur south of Comanche Point and east of Caliente, which were considered the range limits in 1987 according to the 5-Year Review (USFWS 2011).

Natural History

Habitat Requirements

Bakersfield cactus grows primarily in chenopod scrub, but is also found in valley and foothill grassland; and occasionally in cismontane woodland, including blue oak woodland and riparian woodland (CNPS

2011; USFWS 2011; CDFW 2013a; Jepson Flora Project 2011). Some associated species include California filago (*Filago californica*), yellow pincushion (*Chaenactis glabriuscula*), and red brome (*Bromus madritensis* ssp. *rubens*), as well as other non-native annual grasses (USFWS 2011).

Bakersfield cactus occurs on floodplains, ridges, bluffs and low rolling hills, and flats (USFWS 2011; CDFW 2013a). Soils are sandy or gravelly with little silt and clay, are low in organic matter, and may contain cobbles or boulders (CNPS 2011; USFWS 2011); they are granitic and well-drained (CDFW 2013a). Bakersfield cactus ranges from 90 meters (295 feet) (CNPS 2011; CDFW 2013a) to 5,000 feet (Kentner, pers. comm. 2012). Table 1 lists primary habitat associations and parameters for Bakersfield cactus.

Table 1. Habitat Associations for Bakersfield Cactus

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Saltbush scrub, grassland, blue oak woodland, and riparian woodland	Primary	Coarse well-drained sandy or gravelly soils, from 90 to 1,140 meters (295 to 5,000 feet) elevation	CNPS 2011; CDFW 2013a; USFWS 2011

Reproduction

Bakersfield cactus blooms from April to May (CNPS 2011).

The pollination biology of Bakersfield cactus is only relevant for the portion of the population that is genetically capable of reproduction by seed. However, that proportion remains unknown (Kentner, pers. comm. 2012). Bakersfield cactus exhibit several features that are characteristic of bee pollination: flowers are large and showy with a watermelon-like odor; it has a long flowering period; and produces large amounts of nutritious pollen from numerous stamens (Jepson Flora Project 2011; Grant and Grant 1979). Flowers of beavertail prickly-pear (*Opuntia basilaris* var. *basilaris*) are commonly visited by

beetles and bees, but are pollinated mainly by bees (Grant and Grant 1979). The native solitary bee, *Diadasia australis* ssp. *california,* is a potential pollinator of Bakersfield cactus (USFWS 2011). This bee is known to occur in Kern County and specializes in collecting pollen from prickly-pear species. *Diadasia* bees in general are oligolectic (exhibit a narrow, specialized preference for pollen sources), with some specializing on cactus species. The little cactus bee (*Diadasia rinconis*) has been recorded as a visitor to Bakersfield cactus (Grant and Grant 1979).

Chromosome counts indicate that at least some Bakersfield cactus are triploid (2 of the 3 plants that have been examined were triploid ($2n$ = 3X = 33); Pinkava et al. 1977, 1992). Triploid plants are typically at least partially sterile and may have a greatly reduced capacity for sexual reproduction either via pollen or by seed. Triploid populations therefore often rely predominantly on vegetative reproduction—the production of new plants from sources other than seed. Fallen pads can take root. Cactus pads may be dispersed by flood waters. Seed dispersal agents are unknown (USFWS 2011), but the fruits and vegetative parts of *Opuntia* species in general, such as the spiny pad, are closely linked with seed dispersal and vegetative dissemination by animals (Reyes-Agüero et al. 2006). Bakersfield cactus does not survive prolonged inundation (USFWS 2011).

Morphological evidence indicates that gene flow (*i.e.* hybridization) between *O. b. basilaris* and *O. b. treleasei* may be occurring in the populations near Oak Creek. The issue of the ploidy of Bakersfield cactus is highly relevant to the question of hybridization between the varieties. Both the proportion of triploid vs. diploid individuals in Bakersfield cactus populations and the frequency with which triploid individuals produce euploid gametes that would be compatible with the gametes of diploid individuals, including *O. b. basilaris*, is currently unknown (Pinkava et al. 1977, 1992).

Ecological Relationships

Competition with non-native grasses for water is likely the cause of the decline in the number of cactus pads and low rates of reproduction observed in recent population studies at Sand Ridge Preserve (USFWS 2011).

A study conducted from 2002 to 2005 at Sand Ridge Preserve analyzed the effects of grass clipping and Fusilade II (a grass-specific herbicide) treatments on Bakersfield cactus survival, flower production, and recruitment. Bakersfield cactus declined on the control plots, and the rates of both vegetative and sexual reproduction were low, likely due to a reduction in soil moisture storage by non-native annual grasses in years with below average precipitation. In contrast to the control plots, the number of cactus pads in the clipped plots and herbicide-treated plots increased (USFWS 2011). A decline in pollinators may be partly responsible for the low levels and infrequency of seed set observed (USFWS 2011).

Predation of Bakersfield cactus is unknown, though it is not considered to a threat to this species (USFWS 2011). In Mexico, the seed and fruits of other *Opuntia* species are consumed primarily by rodents, but also by harvester ants, birds, and other mammals (González-Espinosa and Quintana-Ascencio 1986).

Population Status and Trends

Global: G5T2, variety is Imperiled (NatureServe 2011, Conservation Status last reviewed 1990)
State: S2.1, Imperiled (CDFW 2013b)

Once likely more or less continuous east of Bakersfield, the current range of Bakersfield cactus consists of scattered fragments of these once larger populations (USFWS 2011).

Though the total population of Bakersfield cactus was not estimated historically, densely spaced clumps of cactus once covered an estimated area of 2 square miles from the Caliente Creek floodplain onto Sand Ridge (USFWS 2011). When known sites were inventoried in 1989, fewer than 20,000 clumps of Bakersfield cactus were estimated to remain. Only four areas had populations of 1,000 clumps or more: Comanche Point, Kern Bluff, Sand Ridge, and the area north of Wheeler Ridge (USFWS 2011). A status survey in 2010 and 2011 was conducted to determine the current state of the historical occurrences of Bakersfield cactus throughout its range (USFWS 2011; Cypher et al. 2011a). Based on these surveys which focused on existing CNDDB occurrences, 25 occurrences are confirmed extant, 11

are believed to be extirpated, the status of 3 could not be determined, 2 previously unreported populations were documented, and 6 undocumented translocated populations were identified. Therefore, there is a minimum of 33 extant occurrences (Cypher et al. 2011a).

Threats and Environmental Stressors

Agricultural land conversion, oil development, sand mining, urbanization, off-road vehicle use, proposed flood control basins, telecommunication and electrical lines construction, and possibly wildfires were considered threats to Bakersfield cactus habitat at the time of its listing in 1990 (USFWS 2011). Currently, the loss and modification of habitat from agricultural conversion, wind energy development, and urban, especially residential, development remain the largest threats to Bakersfield cactus (USFWS 2011; Kentner, pers. comm. 2012). Threats today also include oil development, off-road vehicle use, sand mining, and competition from non-native grasses. In addition, climate change, air pollution (including elevated nitrogen deposition), loss of pollinators, flooding, and loss of genetic diversity have been identified as potential new threats (USFWS 2011). However, loss of genetic diversity is not relevant to the unknown proportion of the population that is triploid and undergoing clonal reproduction (Kentner, pers. comm. 2012).

Conservation and Management Activities

A recently-completed survey has provided updated information on the status of known occurrences, confirming at least 33 current occurrences (Cypher et al. 2011a).

In 1990, The Nature Conservancy doubled the size of the Sand Ridge Preserve to 270 acres by acquiring a remnant of the Caliente Creek wash at the eastern base of the ridge. In 1997, the preserve was transferred to the Center for Natural Lands Management (USFWS 2011; CNLM 2011).

Since 1993, with implementation of the Metropolitan Bakersfield Habitat Conservation Plan, several colonies of Bakersfield cactus have been acquired. The Implementation Trust for the Metropolitan Bakersfield Habitat Conservation Plan has protected parts of

occurrences within the Kern Bluffs and Sand Ridge recovery sites (USFWS 2011). Negotiations over the proposed Department of Water Resources (DWR) Habitat Conservation Plan (HCP) for the California Aqueduct right-of-way are currently stalled with no target date for HCP completion (Grunewald 2011).

The approximately 100,000-acre Wind Wolves Preserve at the very southern end of the San Joaquin Valley is owned and run by the Wildlands Conservancy. There are approximately 50 acres of presumed occupied Bakersfield cactus habitat on the Wind Wolves Preserve within the Wheeler Ridge recovery site (USFWS 2011).

Tejon Ranch Corporation negotiated with national conservation groups on a preservation agreement, executed on June 17, 2008, in which Tejon Ranch Corporation committed to placing aside 178,000 acres through a combination of dedicated and designated project open spaces and allowing the conservation organizations to purchase up to an additional 62,000 acres at State-appraised cost. The conservation easement established through the agreement would result in the permanent conservation of almost 90% of the Ranch (USFWS 2011).

The California Native Plant Society (CNPS) transplanted Bakersfield cactus clumps from sites proposed for development to Sand Ridge Preserve and the California Living Museum in Bakersfield. In addition, a few of the cactus clumps growing on the East Hills Mall site in Bakersfield were removed prior to mall construction, then replanted in a display bed after construction. No monitoring of transplanted individuals has occurred at any of the sites to determine survival rates or reproductive success (USFWS 1998). Hundreds if not thousands of Bakersfield cactus plants have been relocated during the construction of wind energy developments near Oak Creek and Mojave. Relocations there are ongoing (Kentner, pers. comm. 2012).

Data Characterization

Distribution of Bakersfield cactus is not well known. It likely occurs in additional locations that have not been documented considering there is a lot of potential habitat that has not been surveyed, primarily because this habitat occurs on private land (Cypher et al.

2011). The recent expansion on the range to include the eastern Tehachapi Mountains from recent occurrences found on wind energy development project sites in Oak Creek and Mojave has not become publically available through the CNDDB at this time (Kentner, pers. comm. 2012).

Although inferences can be made from other *Opuntia* species, the reproductive biology of Bakersfield cactus has not been studied directly (USFWS 2011).

Management and Monitoring Considerations

The USFWS 5-year review identified the following five highest priority actions to be implemented over the next 5 years to achieve progress toward recovery (USFWS 2011):

1. Protect populations within Bakersfield City limits in the Kern Bluff area and south of Highway 178
2. Work with willing landowners to establish a conservation easement or fee title to the property at the mouth of Kern Canyon
3. Complete the draft Department of Water Resources Habitat Conservation Plan
4. Conduct census of known populations and monitor the reproductive status of known populations
5. Determine suitable management methods for reducing non-native annual grasses and increasing native perennials, including Bakersfield cactus, and communicate the benefits of such management to rangeland landowners.

Cypher et al. (2011b) translocated Bakersfield cactus pads and clumps from the Center for Natural Land Management's Sand Ridge Preserve to Kern County's Bena Landfill Conservation Area as part of a trail population establishment. Ten clumps and 25 shed pads were translocated in fall 2009. Cypher et al. (2011b) concludes that translocation may constitute an effective strategy for establishing new populations of Bakersfield cactus, but suggests continued monitoring of the success of the Bena Landfill population.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Bakersfield cactus, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are approximately 3,421 acres of modeled suitable habitat for Bakersfield cactus in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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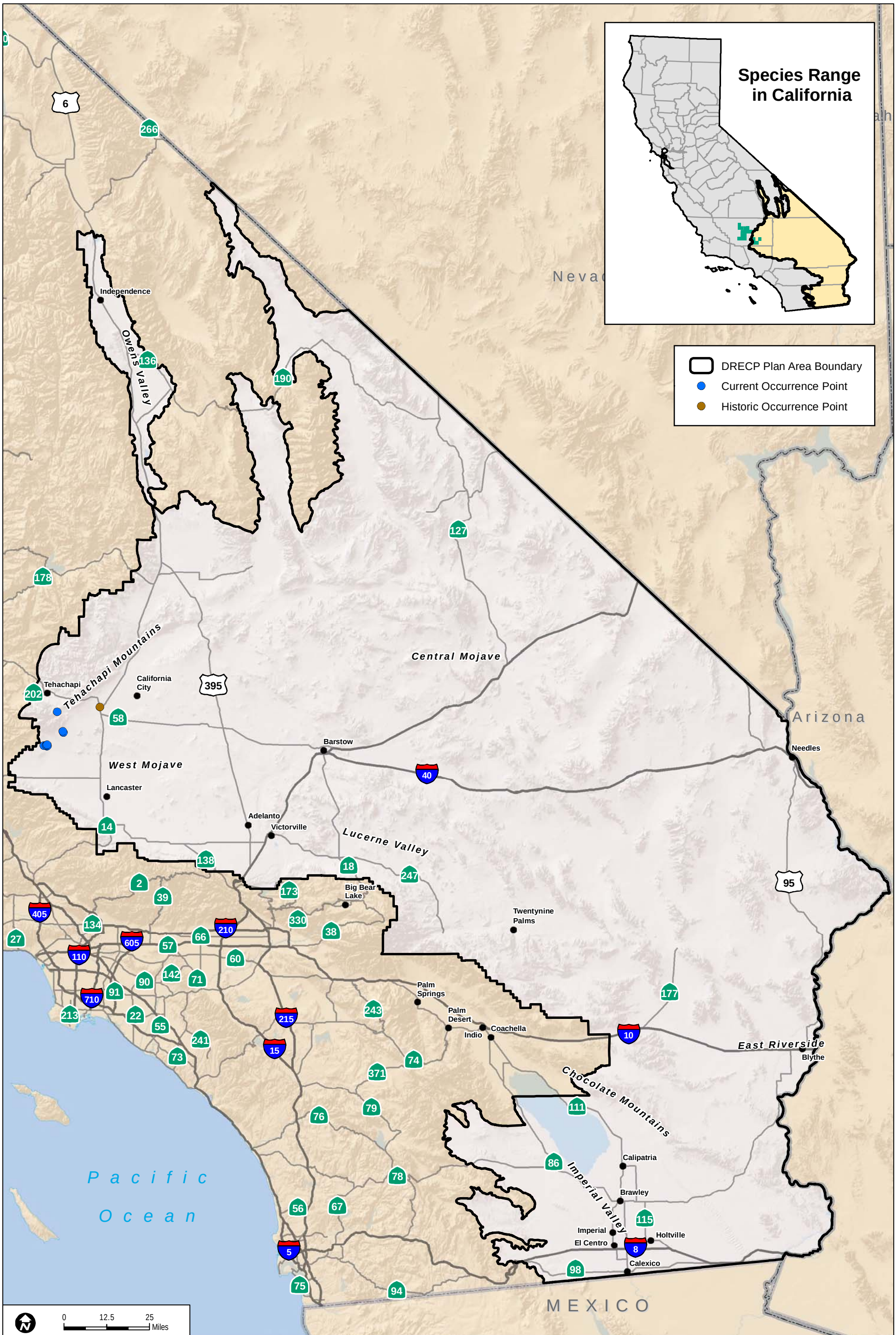

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P02

Bakersfield Cactus Occurrences in the Plan Area

Barstow woolly sunflower (*Eriophyllum mohavense*)

Photo courtesy of Xeric Specialties.

Legal Status

State: None
California Rare Plant Rank: 1B.2[1]
Federal: Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A
Notes: In 1993, the U.S. Fish and Wildlife Service (USFWS) determined that proposing to list Barstow woolly sunflower as endangered or threatened may have been appropriate, but sufficient data on biological vulnerability and threat were not available at that time to support a proposed rule (58 FR 51144–51199).

Taxonomy

Barstow woolly sunflower (*Eriophyllum mohavense*) was originally described by Ivan Murray Johnston in 1923 under the synonym *Eremonanus mohavensis* (Johnston 1923; IPNI 2005), but soon included in Eriophyllum by Jepson (1925, p. 1117). Barstow woolly sunflower is in the sunflower family (Asteraceae) (Jepson Flora Project 2011). It is an annual herb standing approximately 1 to 2.5 centimeters (0.4 to 1 inch) in height. A full physical description of the species can be found in The Jepson Flora Project (2011) and Munz (1974).

Distribution

General

This species is endemic to California's Mojave Desert (Jepson Flora Project 2011). Barstow woolly sunflower is restricted to a range within a 30-mile radius of Kramer Junction in San Bernardino and

[1] **1B:** Rare, threatened, or endangered in California and elsewhere; **X.2:** Fairly threatened in California.

Kern Counties. The eastern-most extant location is Barstow, while the westernmost is the town of Mojave, southernmost is El Mirage, and the northernmost is 25.8 mi northeast of Kramer Junction between Almond Mountain and Black Hills (CDFW 2013a). The species' elevation range extends from 2,000 to 3,600 feet (CDFW 2013a). All of the 67 total California Natural Diversity Database (CNDDB) occurrences (at 168 localities) are in the Plan Area (Figure SP-P04).

Distribution and Occurrences within the Plan Area

Historical

There are 168 total CNDDB localities in the Plan Area, approximately 22% (37) of which have been recorded prior to 1990 (CDFW 2013a). Additional occurrences of Barstow woolly sunflower have been extirpated without having been updated in the CNDDB (MacKay, pers. comm. 2012). The historic occurrences extend from the area around Barstow northwest to the Almond Mountains foothills, west to the area around Kramer Junction, and south to Stoddard Mountain (CDFW 2013a).

Recent

The majority of the 134 CNDDB localities recorded since 1990 are located in the vicinity of Kramer Junction on Edwards Air Force Base. Known extant occurrences now extend farther west, approximately 5.5 miles east of the Mojave Airport, and near Buckhorn Lake about 1 mile north of the Kern–Los Angeles County line. New records farther east are from near Opal and Lane Mountains, as well as Barstow (Figure SP-P04). The El Mirage CNDDB occurrence, entered in November 2011, is now the known southernmost occurrence. Of the current localities, approximately 30% are on lands owned by the Department of Defense (DOD) on Edwards Air Force Base, 10% are on Bureau of Land Management (BLM) land, 6% are on lands managed by the CDFW in the West Mojave Desert, and 54% are on lands that are privately owned or are likely privately owned (CDFW 2013a).

Natural History

Habitat Requirements

Barstow woolly sunflower has been observed in openings within chenopod scrub, Mojavean desert scrub, creosote bush scrub, and also occurs on playas (CNPS 2011; Jepson Flora Project 2011). This species has been observed on bare areas with little soil that frequently contain a shallow subsurface caliche layer (BLM 2005) (Table 1). Barstow woolly sunflower often grows in the sandy margins of small "scalds", which are slightly depressed areas (within the preferred vegetation types) with poor drainage that collect water and then evaporate. However, further away from the Kramer Junction/ Edwards Air Force Base areas, it has been reported growing under different edaphic conditions. For example, the easternmost CNDDB location is on a cobbly ridge, north-facing slope, and the occurrence at Opal Mountain is on upland gravelly soil (CDFW 2013a; MacKay, pers. comm. 2012). A 1995 study by the consulting firm, TetraTech, showed that this species tends to occupy soils with more clay in upper layers, higher alkalinity, more boron, and soil of harder consistency than adjacent unoccupied areas (cited in Andre).

Table 1. Habitat Associations for Barstow Woolly Sunflower

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Chenopod scrub, Mojavean desert scrub, Creosote bush scrub, and Playas	Primary habitat	2,000–3,600 feet	CNPS 2011; CDFW 2013a

Reproduction

Barstow woolly sunflower is a very small annual plant. Duration of flowering is from two to three weeks during the flowering period from March or April to May. Plants then generally go to fruit in May (CNPS 2011; Jepson Flora Project 2011). An 8-year study by Jim Andre in the 1990s showed that seedlings are only established in years of

average or above average precipitation. The study also showed that populations tend to occupy the same places when they do germinate, possibly indicating that there is very limited seed dispersal distance in this species (Andre and Knight 1999). Plants were successfully established off site as mitigation for the Luz solar field project (MacKay, pers. comm. 2012). There is no information available regarding pollinators.

Ecological Relationships

Very little is known about the ecological relationships of Barstow woolly sunflower. Annual species that are most frequently found with Barstow woolly sunflower in the same microhabitat include Mojave spineflower (*Chorizanthe spinosa*) and yellow pepper-grass (*Lepidium flavum*). Mojave spineflower was reported as an associated species is over half of the CNDDB records and yellow pepper-grass was an associated species in several records as well (CDFW 2013a).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed 2006)
State: S2.2, Imperiled (CDFW 2013b)

The 2012 CNDDB includes 63 occurrences for this species, although this estimate includes occurrences that are historic (prior to 1990) or possibly extirpated (CDFW 2013a). Population trends for this species are unknown at this time, but a multi-year, population-level study is underway by BMP Ecosciences and estimated to conclude in 2015. This is an annual plant with populations that fluctuate greatly (by orders of magnitude) from year to year depending on conditions, and also which have a soil seed bank that also likely shows a remarkable amount of fluctuation. Barstow woolly sunflower responds to water availability in terms of population dynamics (Andre and Knight 1999).

Threats and Environmental Stressors

Threats to Barstow woolly sunflower include military activities, energy and subdivision development, sheep grazing, exotic plant species, off-road vehicle use, highway and road improvements and

building, mining, dumping, and pipeline construction (NatureServe 2010; CNPS 2011; MacKay, pers. comm. 2012). Of these threats, those of primary concern include energy development, military activities, sheep grazing, off-road vehicles, and highway improvements (NatureServe 2010; MacKay, pers. comm. 2012). Energy development includes not only construction of solar and wind power production sites, but also utility corridor construction (e.g., roads, transmission lines) (MacKay, pers. comm. 2012).

Specific effects of energy development include shading from solar panels. Shading can reduce the density of Barstow woolly sunflower by suppressing emergence from the seed bank. In addition, shading from solar panels may kill plants before they flower, thus reducing seed production (Tanner et al. 2014). Shading from solar panels can also decrease species richness (i.e., the number of different species present) and community abundance (i.e., the number of individual plants present) (Tanner et al. 2014).

Several Barstow woolly sunflower sites may be extirpated, but their status has not been reported to the CNDDB; however, it is also important to recognize that these plants may be inactive in some years but persist in the seed bank. Currently, only one CNDDB occurrence is recorded as possibly extirpated (CDFW 2013a). However, CNDDB Occurrences #9 and #10 occur along Highway 58 and a widening project has occurred along this highway that has likely extirpated these occurrences (CDFW 2013a; MacKay, pers. comm. 2012).

Conservation and Management Activities

The BLM has established a 314-acre botanical Area of Critical Environmental Concern (ACEC) northeast of Kramer Junction to protect the Barstow woolly sunflower in the West Mojave Plan Area. In a final West Mojave Plan EIS (BLM 2005), of which Alternative A was adopted by BLM in a March 13, 2006 Record of Decision, the protected area for Barstow Woolly Sunflower was expanded to 36,211 acres. This includes the original 314-acre fenced area (now officially called the Barstow Woolly Sunflower ACEC) plus some adjacent CDFG land (acquired by a land exchange with BLM). Along with some

private inholdings, the entire 36,211 acres makes up the Barstow Woolly Sunflower Conservation Area (BLM 2005; MacKay, pers. comm. 2012). This ACEC has a perimeter fence that offers protection from human impacts. However, the BLM has little staff to police and enforce the area, so it is unclear how much protection the Barstow Woolly Sunflower Conservation Area affords this species (MacKay, pers. comm. 2012).

Management areas at Haystack Butte and Leuhman Ridge on Edwards Air Force Base support Barstow woolly sunflower. Another management area consisting of undeveloped land north of Mercury Boulevard also supports this species (Edwards Air Force Base 2002).

Data Characterization

Little is known about the population status and ecology of Barstow woolly sunflower due to its ephemeral life history. Many of the occurrence points are relatively old and need to be updated (MacKay, pers. comm. 2012). Nearly half (29 of 63) of the CNDDB occurrences were recorded prior to 1990 or are not dated (CDFW 2013a).

Surveys seem only to be done around existing roads and trails, and especially in areas where there are proposed projects. Much more can be discovered by extensive and thorough surveys on public lands, as well as private lands (if permission granted), conducted within the flowering period and in years with average to above-average precipitation.

Management and Monitoring Considerations

Barstow woolly sunflower would likely benefit from the elimination of off-road vehicle use and sheep grazing in occupied areas. In addition, vast areas remain unsurveyed (MacKay, pers. comm. 2012). Focused surveys for this species should be conducted in suitable habitat where it is likely to occur, including investigating the status of records of the species where the status is uncertain and that may have been extirpated. Management and monitoring are complicated by the year-to-year fluctuations in population size in response to rainfall. It is very important that surveys be during the short flowering season (before fruiting) in years of average to above-average rainfall. The inadequacy

of survey efforts is substantiated by the very recent 2011 discovery of Barstow woolly sunflower at El Mirage (MacKay, pers. comm. 2012).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Barstow woolly sunflower, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are approximately 186,866 acres of modeled suitable habitat in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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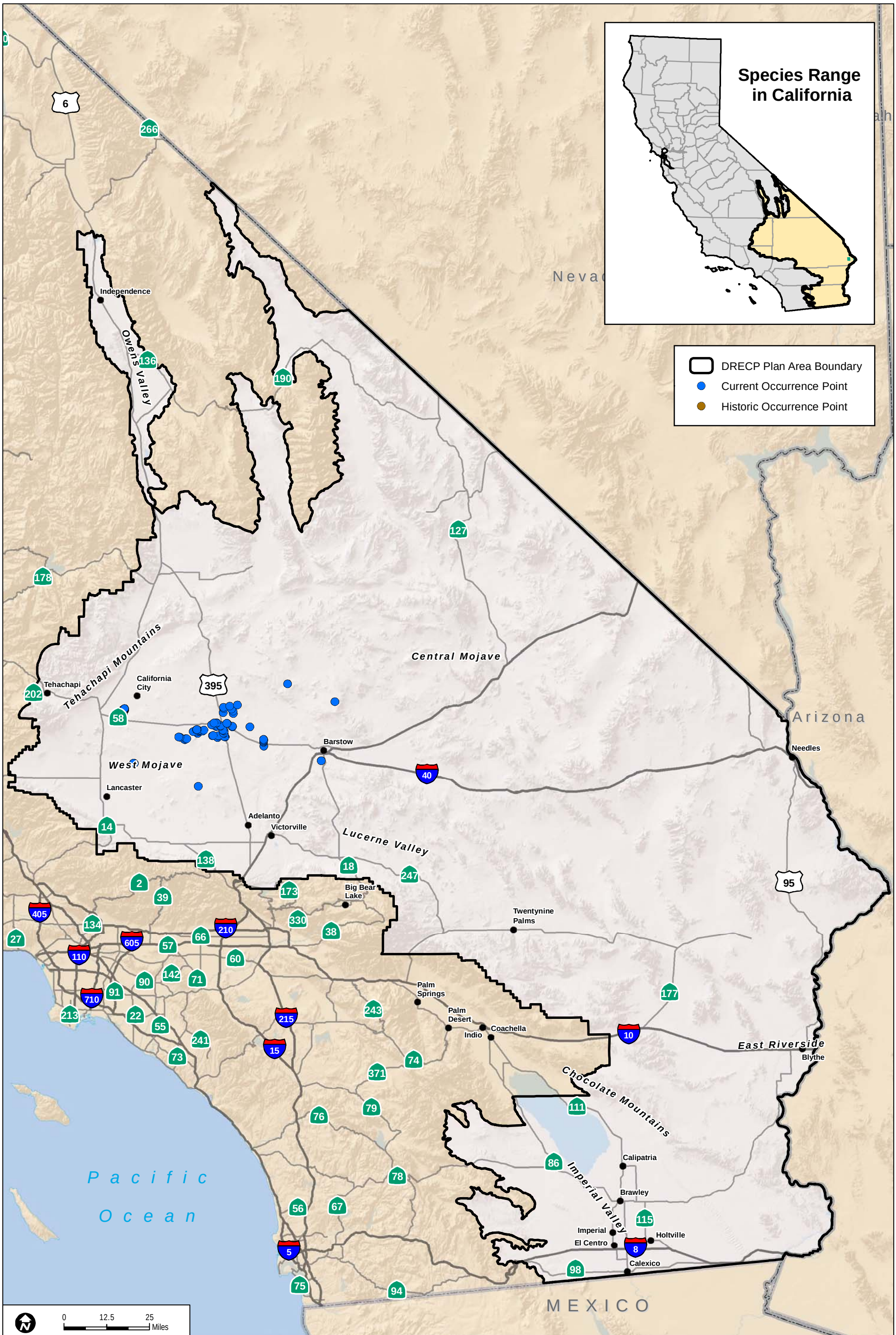

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P03

Barstow Wolly Sunflower Occurrences in the Plan Area

Desert Cymopterus (*Cymopterus deserticola*)

Photo courtesy of Jasmine J. Watts

Legal Status

State: None
California Rare Plant Rank: 1B.2[1]
Federal: Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

Desert cymopterus (*Cymopterus deserticola*) was originally described by Townshend Stith Brandegee in 1915 (Hall 1915, p. 168; IPNI 2005). Mathias (1930) provides a detailed description of this species, and subsequent descriptions in floras appear to be based on this work (Bagley 2006). Desert cymopterus is in the carrot family (Apiaceae) (Jepson Flora Project 2011). Desert cymopterus is a tap-rooted perennial about 15 centimeters (5.9 inches) in height. A full physical description of the species can be found in the Jepson Flora Project (2011).

Distribution

General

There are a total of 79 occurrences in the California Natural Diversity Database (CNDDB) (CDFW 2013a) all originating from 14 collections, one collection of which was a duplicate (Sanders, pers. comm. 2012) The historical distribution of desert cymopterus ranged from Apple Valley in San Bernardino County northward approximately 55 miles to the Cuddeback Lake basin in San Bernardino County, and westward approximately 45 miles to the Rogers and Buckhorn Dry Lake basins on Edwards Air Force Base in Kern and Los Angeles Counties.

[1] **1B:** Rare, threatened, or endangered in California and elsewhere; **X.2:** Fairly endangered in California.

However, the Apple Valley locations have presumably been extirpated resulting in a current distribution that includes the Rogers Dry Lake, Harper Dry Lake, Cuddeback Dry Lake, and Superior Dry Lake basins (69 FR 64884–64889; Figure SP-P06). This species occurs at elevations from 2,000 to 3,000 feet, and possibly up to 5,000 feet (69 FR 64884–64889; CNPS 2011).

Distribution and Occurrences within the Plan Area

Historical

There are three CNDDB occurrences from before 1990. Two of these are located in the vicinity of Leuhman Ridge and Kramer Hills near other occurrences of this species. One of these is possibly extirpated and located over 25 miles southeast of other occurrences east of Victorville (Figure SP-P06; CDFW 2013a).

Recent

There are a total of 230 CNDDB occurrences in the Plan Area (CDFW 2013a). Of these, there are 227 recent occurrences (status updated since 1990) that range from south of Buckhorn Lake along the Kern–Los Angeles County boundary north to the Black Hills and Fort Irwin (Figure SP-P06). However, the majority of these occurrences are located on or near Edwards Air Force Base which may be because Edwards Air Force Base is the only area in the Mojave Desert that has had extensive surveys conducted for desert cymopterus. Those on Edwards Air Force Base and the one occurrence at Fort Irwin are on lands owned by the Department of Defense (DOD). Other occurrences on public land include those managed by the Bureau of Land Management (BLM) in the general vicinity of North Edwards, Harper Lake, and Cuddeback Lake. The remaining nine recent records are either located on private land or the ownership is unknown (CDFW 2013a).

Natural History

Habitat Requirements

Desert cymopterus grows in Joshua tree woodland, saltbush scrub, and Mojavean desert scrub communities on loose, sandy soils. The

sandy soils required by this species occur on alluvial fans and basins, stabilized sand fields, and occasionally sandy slopes of desert dry lake basins (69 FR 64884–64889).

Table 1. Habitat Associations for Desert Cymopterus

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Joshua tree woodland, Saltbush scrub, Mojavean desert scrub	Primary habitat	Loose, sandy soils, 2,000–5,000 feet	69 FR 64884–64889; CNPS 2011

Reproduction

As a taprooted perennial, desert cymopterus does not appear to reproduce vegetatively, but rather reproduces via seeds. Seedling establishment has not been reported for this species. Establishment of new individuals in a population may be infrequent given that many reported desert cymopterus populations are highly dispersed and low density (NatureServe 2010).

Depending on the year, desert cymopterus flowers between early March and mid-May, and may not flower at all in unfavorable years. Poor seed production or seed survival may be a factor in infrequent establishment observed in field studies. At a number of sites in several different years little or no seed production has been observed. A study conducted in 1988 at five sites found that the inflorescences dried up and aborted before setting fruit at each site (Moe 1988, cited in Bagley 2006). In a 1992 study at three sites on Edwards Air Force Base, Charlton (1993, cited in Bagley 2006) reported that only a small portion of the plants flowered and that even fewer successfully produced seed. On the other hand, in 1995, a wet El Niño year, most plants (95%) produced inflorescences at the same three sites, and 51% of the plants had set fruit near the end of the growing season (Mitchell et al. 1995, cited in NatureServe 2010). However, this still indicates a lot of inflorescences aborted before setting fruit (NatureServe 2010).

Fruits of desert cymopterus are fairly large and do not seem well adapted for dispersal over long distances. Fruits generally seem to fall

relatively close to the parent plant. The fruits have a marginal wing that may facilitate dispersal by wind. However, the wings in *C. deserticola* are reduced and appear to be thickened, which suggests that either wind dispersal is less important in this species or that the winds of the Mojave are sufficient to move seeds with poorly developed wings (Sanders, pers. comm. 2012). In addition, the fruits mature late in the season, typically after the end of the rainy season, so they remain dry and light. Therefore, given that wind is relatively common in the open sandy habitats where this species is found, it could easily push the fruits along the soil surface, although the fruits probably do not become airborne (NatureServe 2010).

Because of the annual variability in rainfall, the underground parts of herbaceous desert perennials, including desert cymopterus, must be able to maintain the populations over time with frequent years of reproductive failure; in addition, they must be able to survive prolonged periods of low soil moisture and entire years without aboveground photosynthetic activity (NatureServe 2010).

In dry years, desert cymopterus may not produce flowers or fruit and may even remain dormant underground during the usual growing season. In very wet years, however, they may produce flowers and fruits abundantly. Observations of abundant desert cymopterus in 1995 on Edwards Air Force Base demonstrated the species' ability to survive the 1988–1994 drought in large numbers and with great vigor (NatureServe 2010). Populations of desert cymopterus are probably maintained by periodic recruitment only after years of exceptionally favorable conditions for seed production (Bagley 2006; NatureServe 2010).

Ecological Relationships

Population sizes appear to vary greatly from year to year, evidently in response to the amount and timing of winter and spring rainfall, making it difficult to determine population trends (NatureServe 2010).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed 2005)
State: S2, Imperiled (CDFW 2013b)

Abundance estimates for each population are usually less than 1,000 plants. However, estimating population size is difficult for a number of reasons. First, occurrences and population size fluctuate widely from year to year in response to climatic conditions, especially on the amount of rainfall. Desert cymopterus is dependent upon frequent spring rains. Furthermore, this species may remain dormant underground as a taproot and may not emerge when there is insufficient rainfall, so the number of individuals underground could be greater than the number of individuals aboveground. Also, detectability many be low in years when plants only produce leaves and no inflorescences (NatureServe 2010).

The largest and most robust populations of desert cymopterus occur on Edwards Air Force Base. Seventeen population surveys were performed during a study in 1995, a good year for the species, and population sizes at each location ranged from 1 to 1,929 individuals. In total, 14,093 individuals were counted over an area of 1,465 acres (Tetra Tech 1995, cited in NatureServe 2010).

Threats and Environmental Stressors

Desert cymopterus is potentially threatened by habitat alteration and destruction resulting from military activities on Edwards Air Force Base, the expansion of Fort Irwin, oil and gas development, utility construction, renewable energy development, off-road vehicle use, sheep grazing, Land Tenure Adjustment, and urban development (69 FR 64884–64889; CNPS 2011). However, according to the proposed rule (69 FR 64884–64889), the magnitude and relative importance of most of these potential threats were unknown. Grazing by native and non-native herbivores—presumably including mammals, insects, and desert tortoise (*Gopherus agassizii*)—is also a threat to this species. This may contribute to the low-density, dispersed nature of the majority of reported desert cymopterus populations by limiting the plants' reproductive potential and reducing their vigor (Bagley 2006).

Conservation and Management Activities

The vast majority of plants and acreage of habitat for desert cymopterus are currently thought to occur on the Edwards Air Force

Base. Therefore, this species is not covered by the West Mojave Habitat Conservation Plan (Edwards Air Force Base 2002).

Management areas at Haystack Butte and Leuhman Ridge on Edwards Air Force Base support desert cymopterus. Another management area consisting of undeveloped land north of Mercury Boulevard also supports this species (Edwards Air Force Base 2002). The Edwards Air Force Base Integrate Natural Resources Management Plan offers general conservation measures based on an ecosystem approach with a general goal of conserving and improving the habitat that would benefit all native species (Edwards Air Force Base 2002).

Data Characterization

In general, data availability for desert cymopterus is poor except for population data in some years at Edwards Air Force Base. Population trends are difficult to assess due to the fluctuations caused by variation in rainfall year to year. Furthermore, little is known regarding the species' reproduction, seed dispersal, and recruitment, and nothing is known about pollination. No studies have examined seed viability, longevity in the soil, and predation. Nothing is known of the physiology of dormancy in desert cymopterus or how long plants can survive dormancy. In addition, the requirements for seed germination and establishment of new plants in the population are unknown (NatureServe 2010).

Management and Monitoring Considerations

Protection should focus on currently known to occur on Edwards Air Force Base just south of Rogers Lake, and west and south of Leuhman Ridge. The long-term viability of populations may also rely on the protection of habitat corridors between these populations. Little is known of the distribution and abundance of desert cymopterus off Edwards Air Force Base. Focused surveys for this plant should be conducted in suitable habitat off Edwards Air Force base in favorably wet years to determine if high-density sites exist and how any such areas could be protected (Bagley 2006).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for desert cymopterus, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 344,996 acres of modeled suitable habitat in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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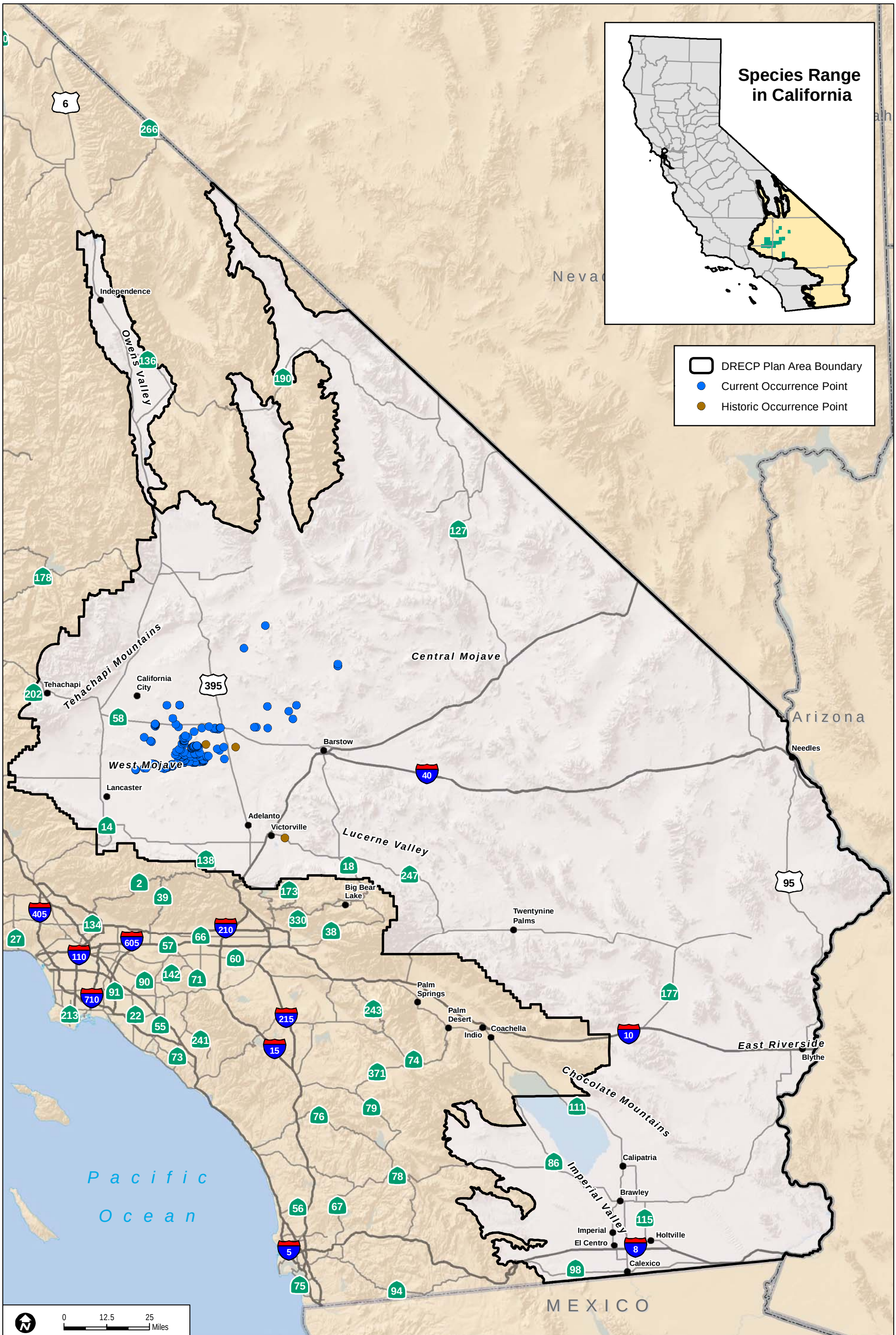

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P04

Desert Cymopterus Occurrences in the Plan Area

Little San Bernardino Mountains Linanthus (*Linanthus maculatus*)

Photo courtesy of Michael Charters, www.calflora.net.

Legal Status

State: S2[1]
California Rare Plant Rank: 1B.2[2]
Federal: Bureau of Land Management Sensitive
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

Little San Bernardino Mountains linanthus (*Linanthus maculatus*) is an annual herb in the phlox family (Polemoniaceae). The species was first described as *Gilia maculata* by S.B. Parish in 1892 from an 1889 collection at "Agua Caliente" (Palm Springs) by W.G. Wright (Jepson Flora Project 2011). During a review of the phlox family in 1904, Milliken treated this species as *Linanthus maculatus* (Milliken 1904) where it remained until the late 1980s. The species has been the subject of much controversy over the last two decades, compounded by a lack of specimens and a lack of close relatives, with Patterson (1989) concluding that the species, although unique, would best fit in the genus *Gilia*, and later Grant (1998) suggesting that the species be placed in the monotypic genus *Maculigilia*. Finally, Porter and Johnson (2000) rebutted Grant's revision and suggested that the species should be returned to the genus *Linanthus*. The taxonomical debate over the placement of this species in *Gilia* or *Linanthus* or some other genus is unlikely to influence its current legal or conservation status.

Little San Bernardino Mountains linanthus is a diminutive, densely hairy, alternate-leaved annual species approximately 1 to 3 centimeters (0.4 to 1.2 inches) in height (Jepson Flora Project 2011;

1 **S2:** Imperiled.
2 **1B:** Rare, threatened, or endangered in California and elsewhere; **.2:** fairly threatened in California.

Patterson 1989). Descriptions of the species' physical characteristics can be found in the *Jepson eFlora* (Jepson Flora Project 2011) and in Patterson's (1989) taxonomic review of the species.

Distribution

General

Little San Bernardino Mountains linanthus is endemic to Southern California with occurrences in San Bernardino, Riverside, and Imperial counties (CNPS 2011). There are 35 collections of Little San Bernardino Mountains linanthus listed in the Consortium of California Herbaria (CCH) database (CCH 2011). The California Natural Diversity Database (CNDDB) records 39 occurrences for this species at 53 localities, but only 27 occurrences at 29 localities occur within the Plan Area (CDFW 2013a). This species' range is restricted to the mouth of Dry Morongo Canyon near the City of Desert Hot Springs and the north side of Joshua Tree National Park south of State Highway 62 in the Little San Bernardino Mountains, and from Whitewater Canyon in the eastern San Bernardino Mountains to Palm Springs. Virtually all of the Palm Springs populations are considered extirpated due to development (Sanders 2006). Additional areas where the species has been recently documented include the mouth of Rattlesnake Canyon and near the Two Hole Spring area on the northern side of the San Bernardino Mountains, and just east of the San Diego County line near Dos Cabezas Spring in Imperial County (Figure SP-P09) (CCH 2011; Sanders 2006).

Distribution and Occurrences within the Plan Area

Historical

Of the 29 localities documented in the CNDDB within the Plan Area, two localities east of Yucca Valley and west of Joshua Tree in San Bernardino County, California, is considered historical because the plants were observed once in 1937 and once in 1940, but these two localities are still presumed to be extant (Figure SP-P09) (CDFW 2013a).

Recent

The 27 recent localities of Little San Bernardino Mountains linanthus occur along the western boundary of the Plan Area in San Bernardino and Riverside counties (Figure SP-P09) (CDFW 2013a). Eight of the localities are at least partially located in Joshua Tree National Park. Seven are located on Bureau of Land Management (BLM) land in Johnson Valley, Homestead Valley, or southeastern Lucerne Valley. One is located on BLM land at the northeastern base of the San Bernardino Mountains and another is at the transition between San Bernardino and Little San Bernardino mountains (CDFW 2013a). Three are located on BLM land in Palm Canyon Wash east of San Diego County. Two localities occur on private land south of the town of Joshua Tree. The remaining five localities have unknown ownership and occur on a wash north of Joshua Tree National Park, south of State Route 62 east of Joshua Tree, at Pipes Canyon north of Yucca Valley, around Yucca Valley, and east of Yucca Valley (CDFW 2013a).

Natural History

Habitat Requirements

Little San Bernardino Mountains linanthus grows on loose, well-aerated, open sandy benches and flats on the margins of desert washes (Sanders 2006; Jepson Flora Project 2011). It grows at 195 to 2,075 meters (640 to 6,806 feet) elevation (CDFW 2012b; CNPS 2011). A review of the elevation data from herbarium collections in the CCH (2011) indicates that the elevation range of the species is from 997 to 4,002 feet (one record indicating a collection from 20 meters elevation appears to be erroneous).

Little San Bernardino Mountains linanthus is always found in open areas that receive no shade from nearby shrubs and is associated with other small annual species, such as sigmoid threadplant (*Nemacladus sigmoideus*), blushing threadplant (*N. rubescens*), evening primrose (*Camissonia pallida*), common loeflingia (*Loeflingia squarrosa*), Arizona nest straw (*Filago arizonica*), and Wallace's woolly sunflower (*Eriophyllum wallacei*) (Sanders 2006).

Table 1. Habitat Associations for Little San Bernardino Mountains Linanthus

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Desert wash[3] systems associated with desert dunes, Joshua tree woodland, and Mojavean and Sonoran desert scrub	Primary habitat	Loose sandy soils, 640 to 6,806 feet elevation	Sanders 2006; Jepson Flora Project 2011; CNPS 2011; CDFW 2013b

Reproduction

Little San Bernardino Mountains linanthus is a diminutive herbaceous annual that reproduces via seed. The ecology of Little San Bernardino Mountains linanthus is not well known because it has not been well studied, and little is known about the plant's pollinator relationships, seed viability, or seed germination (Patterson 1989; Sanders 2006; CVAG 2006). The flower is white with a vermillion spot on each spreading lobe on most individuals (Munz 1974), suggesting that the species is almost certainly insect-pollinated (Sanders 2006). The flowering time for this species is March through May (CNPS 2011). A review of the collections shows that approximately one-third of the specimens were collected in March, two-thirds in April, and only a few in February and May (CCH 2011).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed)
State: S2, Imperiled (CDFW 2013b)

There are four major populations of Little San Bernardino Mountains linanthus (Sanders 2006). All populations are extant except for the Palm Springs populations, which were located in the center of what is now Palm Springs and along Interstate 10 north of the city proper (Sanders 2006). Because of the isolated nature of desert wash systems,

[3] Sanders (2006) states that populations are found only on loose sandy benches on the margins of washes… shrubs are always present in the general areas occupied, but these are not common on the sandy benches where *Gilia* actually is found.

the major populations are separated into smaller "population units" associated with individual washes (Sanders 2006). Two new populations have been discovered in the last two decades: a population in the Rattlesnake Canyon and Two Hole Spring areas on the northern side of the San Bernardino Mountains and an Imperial County population located just east of the San Diego County line near Dos Cabezas Spring (CDFW 2013a; CCH 2011).

Some estimates have been made of the number of individuals in some occurrences. About 10,000 individuals were estimated north of Indian Avenue near the mouth of Big Morongo Canyon (Riverside County) in 1996 and widespread plants observed in flat areas between Joshua Tree and Indian Cove in 1995 (G. Hemkamp, pers. comm., cited in Sanders 2006). A few hundred individuals were present in the Dry Morongo Canyon (San Bernardino County) area in 1992 and 1995 and six in 1996; and 100 plants in an area south of Joshua Tree near State Highway 62 in 1986, which were "reduced markedly" in 1987, 150–200 plants in 1988, 25–30 plants in 1990, and 1,000 plants in 1993 (Patterson 1989; CDFW 2013a).

There are several gaps in the early records for this species, including a 17-year gap from 1907 to 1924 (Sanders 2006; CDFW 2013a; CCH 2011). Only six collections were made between 1924 and 1960 and only two collections were made in the 1970s. Since the end of the 1970s, the number of collections has increased, probably because of the increase in desert botanical work and Patterson's 1989 description of habitat for the species (Sanders 2006).

Population trends are difficult to estimate for the species because population size in a given year appears to depend on environmental conditions and fluctuates greatly from year to year.

Threats and Environmental Stressors

Little San Bernardino Mountains linanthus is potentially threatened by habitat disturbance and destruction from urban expansion, off-highway vehicle use, illegal dumping, and an increase in invasive non-native species (CNPS 2011; CDFW 2013b), and flood control activities (CVAG 2006). The largest populations are adjacent to communities, such as Yucca Valley, Joshua Tree, and Desert Hot Springs, that have

grown substantially in the last two decades. Additional development pressures associated with the expansion of these communities could impact core populations (Sanders 2006).

Flood control maintenance activities pose a specific threat to the species as these activities change the hydrological regime and sediment-carrying capacity of flows within wash systems. In particular, flood control activities pose a substantial threat to populations of Little San Bernardino Mountains linanthus in the Whitewater Canyon, Mission Creek, and Dry Morongo Canyon Wash areas (CVAG 2006).

Off-highway vehicle use is a particular threat to Little San Bernardino Mountains linanthus because the species grows only in desert washes, which are favored by off-highway vehicle users because they are so sparsely vegetated (Sanders 2006).

Conservation and Management Activities

The Coachella Valley Multiple Species Habitat Conservation Plan (CV MSHCP) covers the majority of the known extant populations of Little San Bernardino Mountains linanthus. The CV MSHCP identified three "Core Habitat"[4] areas for the species: Whitewater Canyon, Upper Mission Creek/Big Morongo Canyon, and the Morongo Wash Special Provisions Area, as well as two additional areas for conservation (CVAG 2006). Additionally, the CV MSHCP has identified approximately 3,189 acres of potential habitat for Little San Bernardino Mountains linanthus in the CV MSHCP plan area, of which approximately 2,410 acres is identified as Core Habitat. Conservation of Little San Bernardino Mountains linanthus habitat in the CV MSHCP area will amount to 2,955 acres, of which 2,235 acres, or approximately 76%, is identified as core habitat (CVAG 2006).

The CV MSHCP will result in conservation of 97% of the known occurrences of the species in the CV MSHCP plan area. Additionally,

[4] The CV MSHCP defines Core Habitat as "The areas identified in the Plan for a given species that are composed of a habitat patch or aggregation of habitat patches that (1) are of sufficient size to support a self-sustaining population of that species, (2) are not fragmented in a way to cause separation into isolated populations, (3) have functional Essential Ecological Processes, and (4) have effective biological corridors and/or linkages to other habitats, where feasible, to allow gene flow among populations and to promote movement of large predators."

the CV MSHCP has coordinated efforts with the Coachella Valley Flood Control District to ensure that the hydrological regime in the wash systems of conserved areas is maintained to ensure the conservation of core habitat (CVAG 2006).

The BLM West Mojave Plan (WMP) area encompasses the large population of Little San Bernardino Mountains linanthus located along the northern edge of Joshua Tree National Park in the Little San Bernardino Mountains, as well as the newly discovered populations in Rattlesnake Canyon and Two Hole Spring on the northern edge of the San Bernardino Mountains (Sanders 2006). The WMP proposes two goals and two objectives for Little San Bernardino Mountains linanthus. The goals are to: (1) protect all occurrences of the species on public lands and protect 90% of occurrences on private lands, and (2) protect drainages and the fluvial processes that define the hydrologic regimes in the wash systems. WMP objectives are to: (1) declare all occupied habitat within 100 feet of the edge of washes as Conservation Areas, and (2) limit the channelization of occupied washes (BLM 2005).

Data Characterization

Population trends for the species are difficult to determine because it appears that yearly fluctuations in population size are correlated with annual rainfall amounts.

Very little data existed for the species prior to Patterson's 1989 review of the species. Since then, much more information has been gathered and synthesized for the species, especially through the drafting of species accounts and species-specific conservation management plans under the CV MSHCP and the BLM WMP. In addition, many new populations or localities have been discovered and mapped since 1989, resulting in a greater understanding of the prime core habitat parameters for the species. Despite a general lack of knowledge on the ecology of the species (pollinator interactions, seed viability, germination requirements, etc.), it appears that enough data have been gathered to effectively draft conservation and management plans for the species.

Management and Monitoring Considerations

Future management efforts for Little San Bernardino Mountains linanthus should focus on maintaining natural unobstructed hydrological regimes in areas that support existing populations, as well as in areas with prime core habitat. This will undoubtedly entail working closely with local flood control agencies and private landowners. Additionally, increased management of off-highway vehicle use, and stricter penalties for their illegal use in areas known to support Little San Bernardino Mountains linanthus should be considered. Finally, future monitoring efforts should focus on determining population trends for known populations, as well as on identifying locations of new populations.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Little San Bernardino Mountains linanthus, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 343,289 acres of modeled suitable habitat for little San Bernardino Mountains linanthus in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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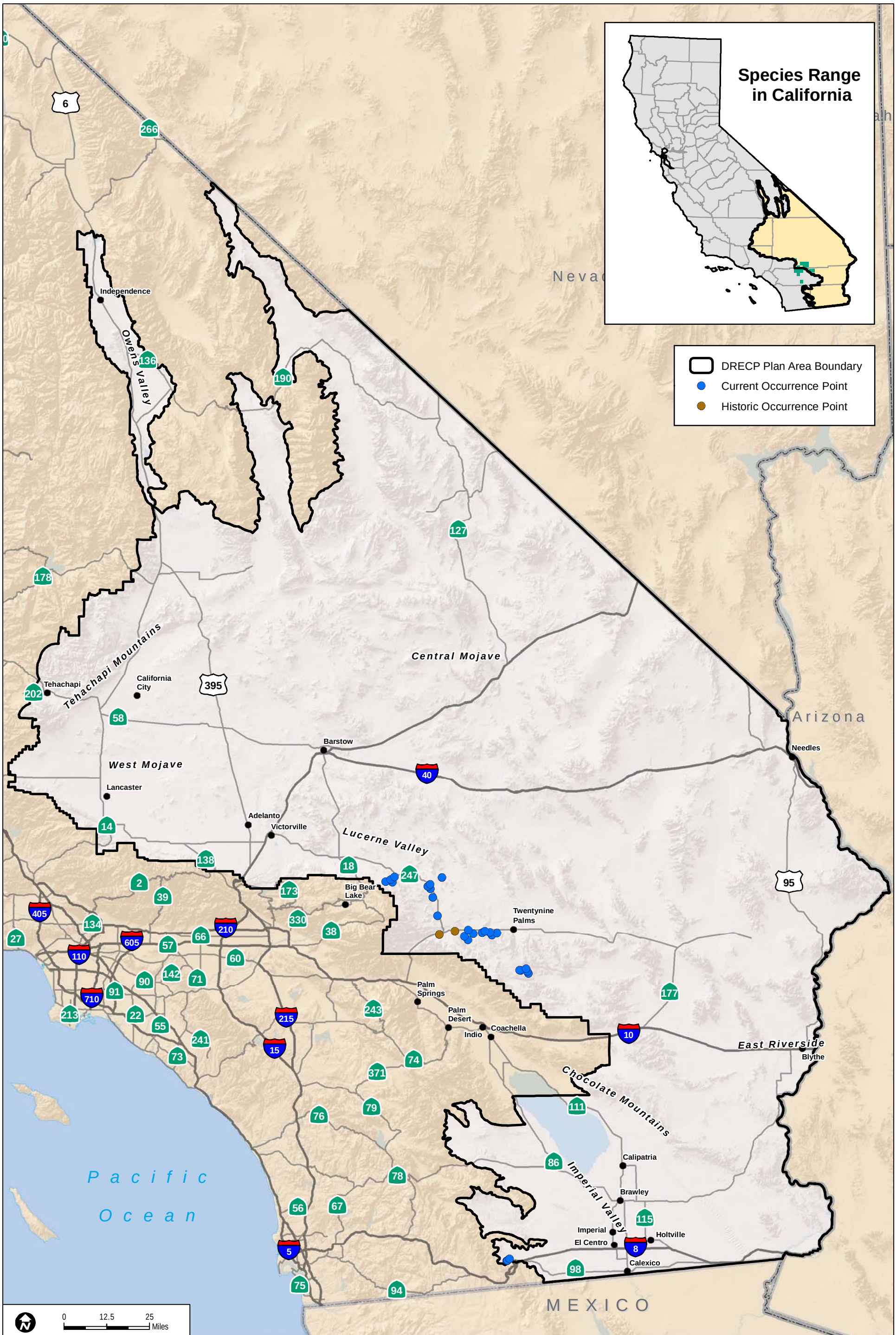

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P05

Little San Bernardino Mountains Linanthus Occurrences in the Plan Area

Mojave Monkeyflower (*Mimulus mohavensis*)

Photo courtesy of Steve Schoenig.

Legal Status

State: None
California Rare Plant Rank: 1B.2[1]
Federal: BLM Sensitive
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

Mojave monkeyflower (*Mimulus mohavensis*) was originally described by John Gill Lemmon in 1884 (Lemmon 1884; IPNI 2011). It is a distinctive member of the genus that was previously placed in its own section (Beardsley et al. 2004). Until recently, Mojave monkeyflower was included in the figwort family (*Scrophulariaceae*), but it is now placed in the lopseed family (*Phrymaceae*) (Beardsley and Olmstead 2002; Jepson Flora Project 2011). There are also current studies that provide evidence that the genus *Mimulus* should be fragmented into several new genera, so more nomenclatural changes can be expected in the near future for this taxon.

Mojave monkeyflower is an annual plant approximately 2 to 10 centimeters (0.8 to 3.9 inches) in size. A full physical description of the species can be found in the Jepson Flora Project (2011).

Distribution

General

This species occurs in the Mojave Desert in west-central San Bernardino County (Jepson Flora Project 2011). The populations with greatest known densities occur south of Daggett and Barstow (MacKay 2006). However, the majority of the historical occurrences

[1] **1B:** Rare, threatened, or endangered in California and elsewhere; X.**2:** Fairly endangered in California.

in the Barstow area have either been extirpated or impacted (CNPS 2011). The elevation range of this species extends from 600 to 1,200 meters (1,969 to 3,937 feet) (CNPS 2011) (Figure SP-P10). There are a total of 56 California Natural Diversity Database (CNDDB) occurrences for Mojave monkeyflower at 121 localities, all of which occur in the Plan Area.

Distribution and Occurrences within the Plan Area

Historical

Eleven localities have not been observed since 1990. Of these, one site at Kane Springs (Element occurrence 6) was visited more recently (in 2011) and no plants were found so it is uncertain whether any plants occur here. However, the Kane Springs resurvey in 2011 with negative results does not mean the plants are not in the vicinity (MacKay, pers. comm. 2012). One occurrence along Camp Road is not dated and no plants were found at this site in 1986 or in 1998. Moore (pers. comm. 2 012) stated that areas off of Camp Rock Road on the smaller BLM roads represent important Mojave monkeyflower habitat because they have very low levels of disturbance in comparison to those on Camp Rock Road. Another historical occurrence is the type locality in Calico and is likely extirpated (CDFW 2013). These records extend from the area around Barstow southeast to the area around the Newberry Mountains, and one occurrence much farther south near Old Woman Springs (Figure SP-P10; CDFW 2013).

Recent

Of the 121 total CNDDB localities in the Plan Area, 110 have been recorded in the CNDDB since 1990 and are presumed extant. One of the major populations of Mojave monkeyflower recorded in the CNDDB since 1990 that is presumed extant is located southeast of Barstow to Ord Mountain. A second concentration of occurrences is located northeast of Adelanto and extends to Helendale. There is an isolated occurrence just south of the Black Mountains summit (Figure SP-P10). However, if the Stoddard Open Off-Highway Vehicle (OHV) area were surveyed there is a high likelihood that Mojave monkeyflower would be documented, providing a continuum of distribution between the two major areas (MacKay, pers. comm. 2012). The disjunct distributions are

the Kane Springs collection east of Rodman (Element occurrence 6) and the Old Woman Springs collection; both areas still need field work (MacKay, pers. comm. 2012).

According to CNDDB records (CDFW2013), of the 47 current occurrences at 110 localities, the vast majority are on lands managed by the Bureau of Land Management (BLM), and the remaining portion are on lands that are privately owned or whose ownership is unknown (CDFW 2013). However, 14 of the 19 occurrences turned in by B. West (BLM employee at the time, 1992) included information that the BLM-owned lands were under consideration for disposal, and BLM subsequently disposed of the land containing four of those occurrences (CDFW 2013; MacKay, pers. comm. 2012). Also, there is a very high probability that the remaining Brisbane Valley is occupied by Mojave monkeyflower (MacKay, pers. comm. 2012).

Natural History

Habitat Requirements

This species occurs in Mojavean desert scrub, specifically creosote bush scrub (MacKay 2006; CNPS 2011). Mojave monkeyflower is associated with the following species or genera, among others: creosote bush (*Larrea tridentata*), desert senna (*Senna armata*), cheese bush (*Ambrosia salsola*), ratany (*Krameria erecta* and *K. bicolor*), chollas (*Cylindropuntia* spp.), burro bush (*Ambrosia dumosa*), prairie-clovers (*Psorothamnus* spp.), Bigelow's monkeyflower (*Mimulus bigelovii*), desert bells (*Phacelia campanularia*), desert fivespot (*Eremalche rotundifolia*), spiny hopsage (*Grayia spinosa*), and desert trumpet (*Eriogonum inflatum* var. *inflatum*) (MacKay 2006; CDFW 2013).

Mojave monkeyflower commonly occurs in areas that are not subject to regular water flow (MacKay 2006). These areas include the gravelly banks of desert washes with granitic soils and rocky slopes above washes, as well as the sandy openings of creosote bush scrub (MacKay 2006).

Table 1. Habitat Associations for Mojave Monkeyflower

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Mojavean desert scrub, Creosote bush scrub	Primary habitat	Granitic soils, 1,968–3,937 feet	MacKay 2006; CNPS 2011; Jepson Flora Project 2011

Reproduction

Germination is probably dependent upon the amount of precipitation, as population sizes can vary substantially from year to year (MacKay 2006).

Most members of the lopseed family are insect pollinated (Beardsley and Olmstead 2002); and given the showy flowers, Mojave monkeyflower pollinators are probably Hymenoptera (bees, wasps, ants, and sawflies) or Lepidoptera (butterflies and moths). MacKay (2006) hypothesized that the white margin of the corolla reflects ultraviolet light, and the maroon veins extending into this margin act as nectar guides to facilitate pollination.

Small seeds and an annual habit suggest that dispersal of Mojave monkeyflower is mostly abiotic (MacKay 2006; NatureServe 2010). For populations located on rocky slopes above washes, it is probable that gravity carries seeds down into the washes and intermittent water flow may carry seeds further down washes. Although biotic vectors of seed transport are unknown, granivorous ants or rodents may transport seeds over short distances and birds may transport seeds longer distances (MacKay 2006).

Ecological Relationships

Although suitable habitat for this species appears to be fairly abundant, it is quite restricted geographically. Population sizes fluctuate substantially from year to year, probably in response to the amount and timing of precipitation; as an annual, germination and establishment are dependent on the timing and amount of spring rains (MacKay 2006; NatureServe 2010). Unknown unusual germination and establishment requirements may account for the considerable variability in population sizes from year to year (MacKay 2006).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed 2006)

State: S2, Imperiled (CDFG 2012b)

Population trends for Mojave monkeyflower are unknown at present, but a multi-year population-level study is underway by BMP Ecosciences (Moore et al.) and expected to be completed by 2015. One CNDDB locality has been possibly extirpated, and the status of 11 of the 121 total CNDDB localities of Mojave monkeyflower in the Plan Area have not been updated since 1990 (CDFW 2013; MacKay 2006).

Threats and Environmental Stressors

Threats to Mojave monkeyflower include development, mining, non-native plants, solar and wind energy projects, grazing, vehicles, and road development (CNPS 2011; NatureServe 2010; MacKay 2006). Additional potential threats include pipeline installation and quarries and test pits adjacent to populations (MacKay 2006). Mojave monkeyflower is also under threat by the potential for the BLM to convert land occupied by this species to private lands, which could then be developed (MacKay 2006; CDFW 2013). The area under consideration for disposal or land exchange is located between Barstow and Victorville (CDFW 2013).

Because population sizes fluctuate considerably annually in response to environmental conditions, Mojave monkeyflower is susceptible to depletion of the seed bank after a series of drought years. In addition, small population sizes increase the risk of inbreeding, which may result in reduced seed set or reduced seed viability (MacKay 2006).

Conservation and Management Activities

The West Mojave Plan designated Mojave monkeyflower conservation areas in the Plan Area as land managed by BLM (BLM 2005). The Brisbane Valley Mojave Monkeyflower Conservation Area is 10,448 acres and the Daggett Ridge Mojave Monkeyflower Conservation Area is 25,351acres (BLM 2006).

Data Characterization

In general, data availability for the Mojave monkeyflower is poor. The pollination ecology of Mojave monkeyflower is unknown (MacKay 2006). This species may have some unusual germination and establishment requirements that are unknown (MacKay 2006). Mojave monkeyflower is also absent from much apparently suitable habitat and remains relatively restricted geographically (MacKay 2006).

The status of many of the recorded populations of Mojave monkeyflower is unknown. Several occurrences documented in the CNDDB may be extirpated but still presumed extant in the database (MacKay 2006). In addition, location data may be inaccurate, especially for older records labeled Barstow; these collections may actually be from the vicinity of Barstow, and not from what is now the town of Barstow (MacKay 2006).

Management and Monitoring Considerations

Protection of the areas where Mojave monkeyflower is known to occur is important to maintain viable populations of the species. The species would likely benefit from the elimination of off-road vehicle use and livestock grazing in occupied areas south of Barstow and Daggett, as well as maintenance of BLM management of lands between the Mojave River and Interstate 15 between Victorville and Barstow. Management and monitoring are complicated by the year-to-year fluctuations in population size in response to rainfall. A very important consideration is to fully understand where populations occur. Vast and thorough surveys should be conducted during the appropriate flowering season in good rainfall years (MacKay, pers. comm. 2012). Confirmation of site occupancy in suitable habitat should be conducted over multiple years before concluding absence. Moore et al. (in prep) found that novel occurrence discoveries in modeled suitable habitat were strongly predicted by the proximity to recent occurrences.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Mojave monkeyflower, using available spatial information and occurrence

information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 176,190 acres of modeled suitable habitat for Mojave monkeyflower in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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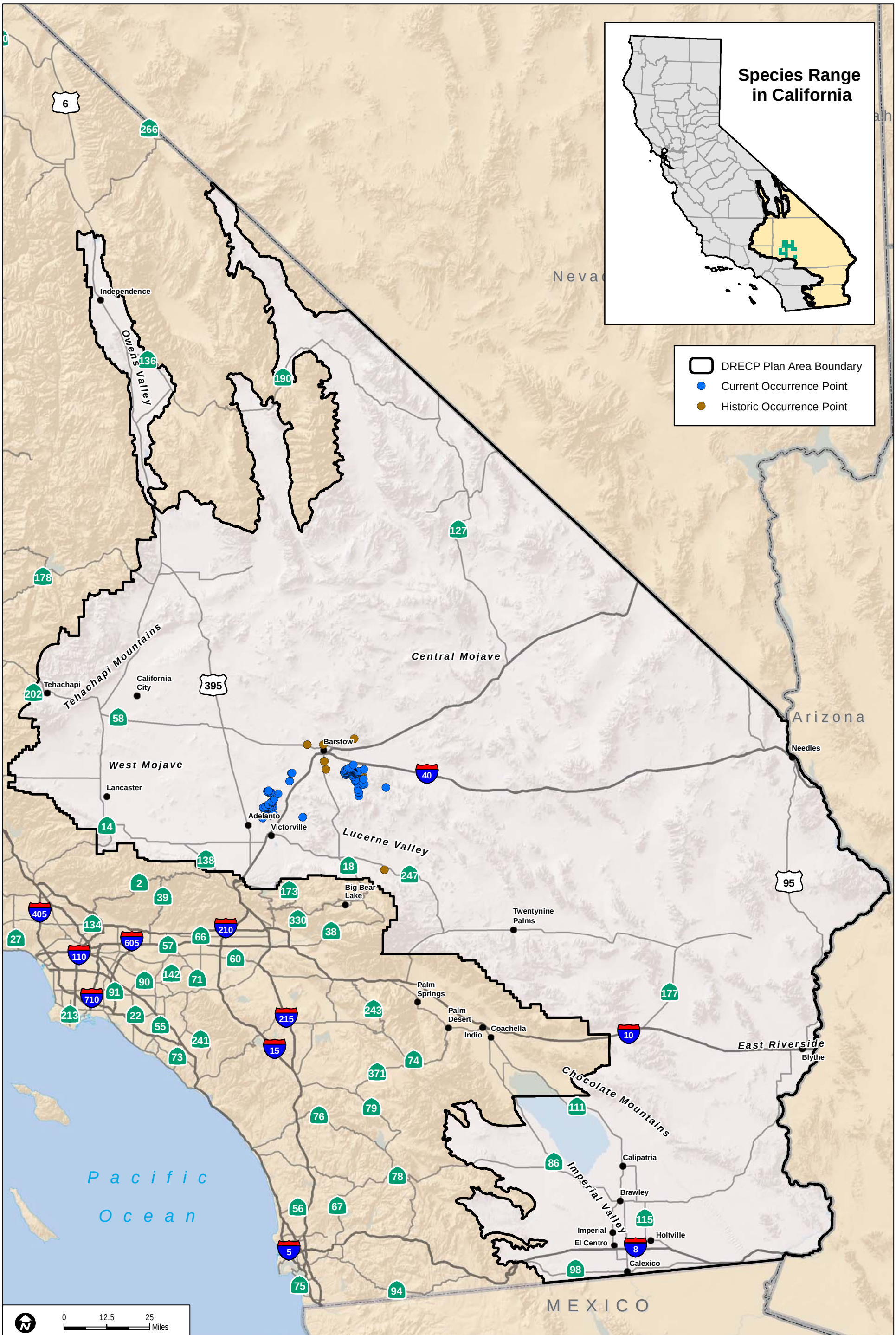

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P06

Mojave Monkeyflower Occurrences in the Plan Area

Mojave Tarplant (*Deinandra mohavensis*)

Photo courtesy of Heath McAllister.

Legal Status

State: Endangered; S2S3[1]
California Rare Plant Rank: 1B.3[2]
Federal: Bureau of Land Management Sensitive; U.S. Forest Service Region 5 Sensitive Plant Species
Critical Habitat: N/A
Recovery Planning: N/A

Taxonomy

Mojave tarplant is in the sunflower family (Asteraceae) (Jepson Flora Project 2011). Mojave tarplant was originally described by D.D. Keck (1935) as *Hemizonia mohavensis* and was reclassified as *Deinandra mohavensis* in 1999 (Baldwin 1999). The taxonomic revision was intended to more accurately reflect phylogenetic relationships within Madiinae (a subtribe within Asteraceae) (Baldwin 1999). The plant was thought to be extinct but was rediscovered by A. Sanders in 1994 in the San Jacinto Mountains, in Riverside County (Sanders et al. 1997).

Mojave tarplant is an annual plant approximately 10 to 100 centimeters (3.9 to 39 inches) in height. A full physical description of the species can be found in the *Jepson eFlora* (Jepson Flora Project 2011).

Distribution

General

There are a total of 75 occurrences in the California Natural Diversity Database (CNDDB) at 124 localities (CDFW 2013a). Mojave tarplant is known in Kern, Riverside, Inyo, and San Diego counties (believed

[1] **S2S3:** the rank is somewhere between S2, Imperiled and S3, Vulnerable.
[2] **1B:** Rare, threatened, or endangered in California and elsewhere; **X.3:** Not very endangered in California.

extirpated from San Bernardino County) (CDFW 2013a) (Figure SP-P11). This species occurs at elevations of 460–1,600 meters (1,509–5,250 feet) (CNPS 2011; Jepson Flora Project 2011). The distribution is discontinuous and possibly relictual.

Distribution and Occurrences within the Plan Area

Historical

This species was not known to occur in the Plan Area prior to 1990 (CDFW 2013a; Figure SP-P11).

Recent

Within the Plan Area, Mojave tarplant is known from the desert slope of the southern Sierra Nevada Mountains in Kern County (Sanders 2006a). There are 10 occurrences at 13 localities in the Plan Area, all within Kern and Inyo counties. The majority of localities are located west of Highway 14 and east of the Sequoia National Forest; north of Interstate 40; near Cutterbank Spring; in Jawbone Canyon; near Short Canyon; in lower Esperanza Canyon; in lower Water Canyon; and in the vicinity of Cross Mountain (CDFW 2013a; Figure SP-P11). Mojave tarplant may also occur at Red Rock Canyon in Red Rock Canyon State Park in Kern County (Faull, pers. comm. 1998, cited in Sanders 2006a).

Natural History

Habitat Requirements

Mojave tarplant occurs in open moist sites in arid regions near the margins of the desert, within chaparral, coastal scrub, desert scrub, riparian scrub, and woodland (CNPS 2011; Sanders 2006a; Jepson Flora Project 2011). Plants are typically observed in seeps and along grassy swales and intermittent creeks. The most suitable habitat occurs in mountainous areas within microhabitats of low gradient streams and on gentle slopes with few shrubs and trees. This species is associated with clay or silty soils that are saturated with water early in the year. Mojave tarplant prefers areas that are dry at the surface but which have a substantial water source at depth through summer.

Dwarfed plants occasionally are found in drier sites near occupied moist areas (Sanders et al. 1997). This cycle of early saturation with later desiccation may reduce competition from other plant species; dryness during drought years may further reduce competition (Sanders 2006a).

At the type locality, Mojave tarplant was known to occur along a sandy intermittent creek; however, this habitat is now believed to be atypical and not suitable to maintain a permanent population. Sanders et al. (1997) note that some occurrences of Mojave tarplant are associated with sand where the sand is adjacent to more typical habitat.

Table 1. Habitat Associations for Mojave Tarplant

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Mesic openings in chaparral, desert and coastal scrub, woodland, and riparian scrub	Primary	Clay or silty soils (sometimes sand); seasonally (winter and spring) saturated with water; 460–1,600 meters (1,509–5,250 feet)	CNPS 2011; Sanders et al. 1997; Sanders 2006a; Jepson Flora Project 2011

Reproduction

Mojave tarplant is an annual plant that blooms from June through January (CNPS 2011). Flowering peaks between August and October. Once flowering has begun, it continues until the plants begin to senesce. Fruit maturity and dispersal are continuous as well. Unlike most former *Hemizonia* species, including the segregated *Deinandra,* Mojave tarplant is self-compatible (Baldwin pers. comm. 1998, cited in Sanders 2006a); the only other self-compatible member of *Deinandra* is Red Rock tarplant (Tanowitz 1982). Pollination studies have not been conducted for Mojave tarplant; however, Faull (1987) observed small beetles and honey bees visiting Red Rock tarplant flowers, a closely related species.

Mojave tarplant blooms from June through January (CNPS 2011). Flowering peaks between August and October. Once flowering has

begun, it continues until the plants begin to senesce. Fruit maturity and dispersal are continuous as well. Seed dispersal vectors have not been reported for this species; however, the seeds are relatively heavy and may just fall to the ground around the source plant. The seeds are not armed with any obvious mechanisms, such as hooks or wings, for long-distance dispersal (Sanders 2006a). Bruce Baldwin (pers. comm., cited in Sanders 2006b) reports that ray achenes of *Hemizonia* (including the segregated *Deinandra*) maintain some degree of dormancy while the disk achenes freely germinate.

Mojave tarplant is known to reproduce easily in cultivation (B. Baldwin, pers. comm. 1998, cited in Sanders 2006a) and has been known to colonize disturbed areas in a botanical garden (S. Boyd, pers. comm. 1998, cited in Sanders 2006a).

Ecological Relationships

As described in Habitat Requirements, Mojave tarplant is associated with seasonally saturated clay or silty soils on gentle slopes or low gradient streams, with few shrubs and trees. These saturated areas are typically dry at the surface but provide a substantial water source at depth through summer (Sanders et al. 1997). This species has a discontinuous and possibly relictual distribution (Sanders 2006a), and little is known of its life history and ecological relationships. Although pollination studies have not been conducted for Mojave tarplant, Faull (1987) has observed small beetles and honey bees visiting Red Rock tarplant flowers, a closely related species. Seed dispersal vectors have not been reported for this species; however, the seeds are relatively heavy and may just fall to the ground around the source plant. The seeds are not armed with any obvious mechanisms, such as hooks or wings, for long-distance dispersal (Sanders 2006a). Mojave tarplant is threatened by grazing, recreational activities, development, hydrological alterations, road maintenance, and vehicles (CNPS 2011). Within the Plan Area, intense cattle grazing and trampling may be the most significant threats.

Population Status and Trends

Global: G2G3, Imperiled/Vulnerable (NatureServe 2011, Conservation Status last reviewed XXXX)
State: S2S3, Imperiled/Vulnerable (CDFW 2013b)

Because this species was only recently rediscovered (in 1994) there is little information available on population trends. Of the 13 occurrences in the Plan Area, four are on BLM lands, two are on private land, and ownership is unknown for two of the occurrences. The occurrence on private land near Cutterbank Spring numbered 14 individuals in 2003. Approximately 15,000 plants were observed at the other occurrence on private land located at the south end of Kelso Valley in 2010. Many more plants were observed in 2011, including an additional 1,500 plants in the northeastern portion of the occurrence (CDFW 2013a). Of the two occurrences for which ownership is unknown, one numbered in the thousands in 1998 and the other numbered 109 individuals in 2003. Of the four occurrences on BLM land, one numbered 50,000 in 2003 (with 30 rosettes observed very early in the year in 2004), one numbered in the several hundreds in 2008, and one numbered 5,000 in 1998 (and was locally common in 2001 and numbered 3,000 in 2003). Approximately 50,000 plants were observed in 2003 at the occurrence at Cutterbank Spring on BLM lands; 30 plants were observed in 2004 in their rosette form in an early season survey, and plants were "abundant around the springs and in the surrounding drainage channels" in 2010 (CDFW 2013a). Overall, there are 69 occurrences in Kern, Riverside, and San Diego counties (CDFW 2013a) and most of these appear to have number of individuals estimated only once, making it difficult to discern a population trend.

Threats and Environmental Stressors

Mojave tarplant is threatened by grazing, recreational activities, development, hydrological alterations, road maintenance, and vehicles (CNPS 2011). The type locality was modified by construction of the Mojave River Forks Dam. Within the Plan Area, cattle grazing occurs at some of the Mojave tarplant occupied areas, and in some areas is locally intense and may pose a threat. However, the sticky plants of the genus *Deinandra* (also called "tarweeds") may not be

palatable to cattle, so grazing may not be a major threat and trampling by cattle around limited watering sources in dry areas may be a greater threat (Sanders 2006a).

Conservation and Management Activities

Four of the occurrences are known from BLM land, two are on private land, and ownership is unknown for two of the occurrences (CDFW 2013a). No current conservation or management activities have been identified for Mojave tarplant.

Data Characterization

The general distribution of Mojave tarplant is discontinuous and patchy. Sanders (2006a) recommends that additional surveys be conducted in the southern Sierra Nevadas and along the north foot of the Transverse Range, particularly the San Gabriel Mountains. Within the Plan Area, four of the occurrences are known from BLM lands, two are on private land, and ownership is unknown for two of the occurrences. Many of the known occurrences outside the Plan Area occur within the San Bernardino and Cleveland National Forests and therefore receive some protection (Sanders 2006a). The species is known to be self-compatible (B. Baldwin, pers. comm. 1998, cited in Sanders 2006a) and a related species (Red Rock tarplant) is known to be insect-pollinated (Faull 1987). Little is known regarding the species' seed dispersal and recruitment.

Management and Monitoring Considerations

Because the global distribution of Mojave tarplant is discontinuous and patchy, Sanders (2006a) recommends that additional surveys be conducted in the southern Sierra Nevadas and along the north foot of the Transverse Range, particularly the San Gabriel Mountains. Additional surveys may identify new occurrences.

Mojave tarplant is threatened by grazing, recreational activities, development, hydrological alterations, road maintenance, and vehicles (CNPS 2011). Measures to control these threats should be considered.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Mojave tarplant, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 270,463 acres of modeled suitable habitat for Mojave tarplant in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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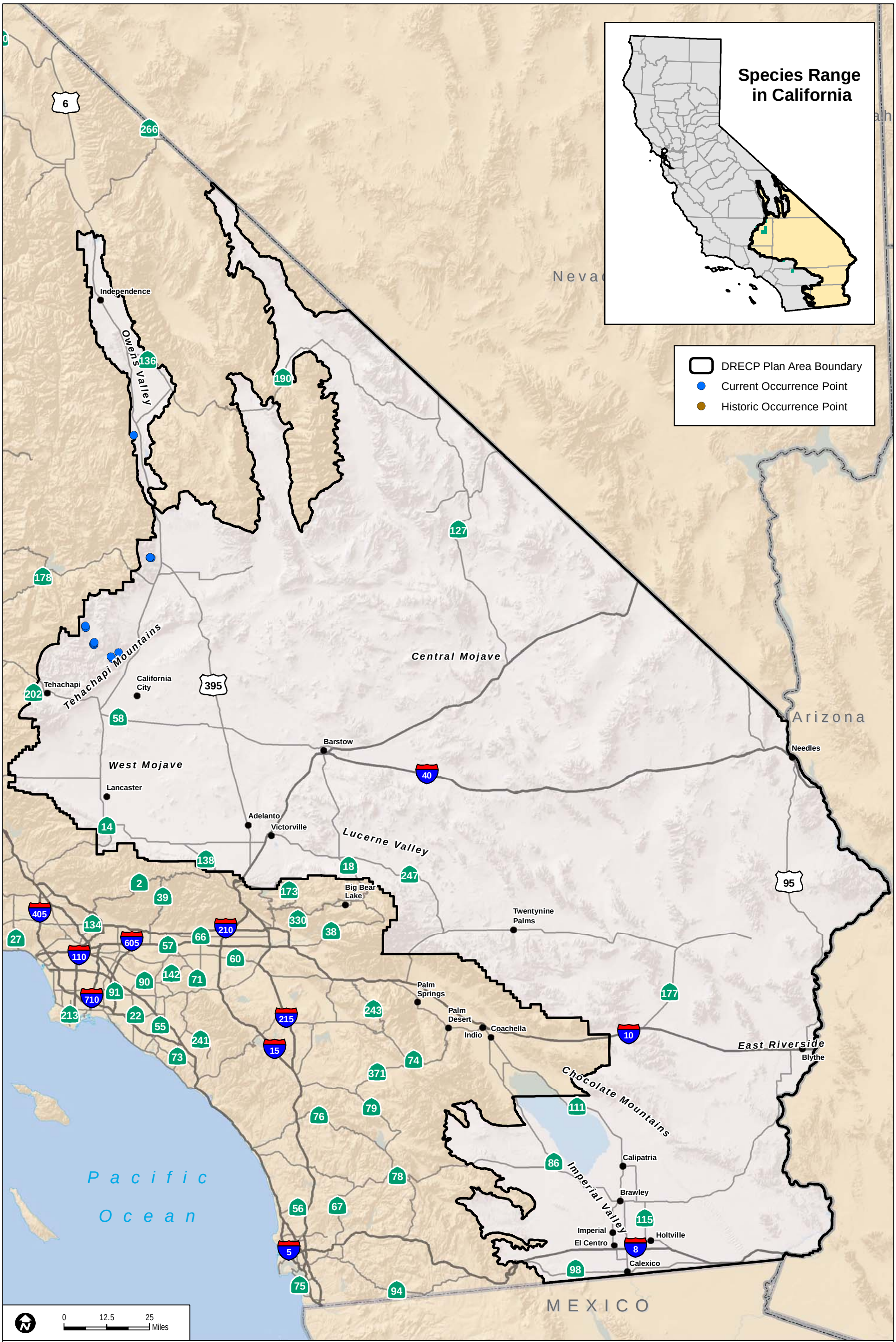

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P07

Mojave Tarplant Occurrences in the Plan Area

Owens Valley Checkerbloom *Sidalcea covillei*

Photo courtesy of Larry Blakely.

Legal Status

State: Endangered; S3[1]
California Rare Plant Rank: 1B.1[2]
Federal: Bureau of Land Management Sensitive
Critical Habitat: None
Recovery Planning: *Owens Basin Wetland and Aquatic Species Recovery Plan, Inyo and Mono Counties, California* (USFWS 2000)
Notes: Considered for federal listing (proposed as a candidate species) in 1985, it was removed from the candidate list in 1996 because the U.S. Fish and Wildlife Service (USFWS) determined that the species was more abundant or widespread than was previously thought, or the species was not subject to any identifiable threat.

Taxonomy

Owens Valley checkerbloom (*Sidalcea covillei*) was originally described by E. Greene in 1914 and the taxonomic status of Owens Valley checkerbloom has not changed since it was first described.

Owens Valley checkerbloom is a perennial herb with stems approximately 2 to 6 decimeters (7.9 to 24 inches) in length. A full physical description of the species can be found in the Jepson Flora Project (2011).

Distribution

General

Owens Valley checkerbloom is endemic to the southern Owens Valley in Inyo County, California (CNPS 2011; BLM 2011b). It grows only in alkali meadow and spring communities scattered along about 125 kilometers (77.7 miles) of the Owens River drainage (Halford 1994). The California

[1] **S3:** Vulnerable.
[2] **1B:** Rare, threatened, or endangered in California and elsewhere; **X.1:** Seriously endangered in California.

Natural Diversity Database (CNDDB) includes 42 occurrences of Owens Valley checkerbloom at 35 localities; 21 of these occurrences are in the Plan Area at 30 localities.

Distribution and Occurrences within the Plan Area

Historical

Owens Valley checkerbloom was first collected in 1891 in an extensive alkali meadow known as Haiwee Meadows, Inyo County, and was not collected again until 1952, when it was found north of Lone Pine in Inyo County. The species was extirpated from its type locality when the Haiwee Reservoir was formed, and by 1978, local botanist Mary DeDecker considered it to be on the brink of extinction (DeDecker 1978). Within the Plan Area, 5 of the 30 known localities are considered historical (i.e., pre-1990) and have not been recently observed. These populations are known to be either extirpated, possibly extirpated, or are presumed to be extant (CDFW 2013a).

Recent

The CNDDB includes 25 recent localities (i.e., since 1990) of Owens Valley checkerbloom in the Plan Area. All of these localities occur on lands owned by the LADWP (CDFW 2013a). All of the localities are generally along Highway 395 from the meadow above Tinemaha Creek south to the area 1 mile north of Olancha (Figure SP-P13; CDFW 2013a).

Natural History

Habitat Associations

Owens Valley checkerbloom grows in moist alkaline meadows and seeps at elevations of 3,580 to 4,650 feet (see Table 1; CNPS 2011; CDFW 2013a). Almost all occurrences grow in fine, sandy loam with alkaline crusts, but one occurrence is known to grow in stony, calcareous soil (CDFW 2013a).

Associated native grasses and herbs include saltgrass (*Distichlis spicata*), alkali sacaton (*Sporobolus airoides*), basin wildrye (*Elymus cinereus*), Baltic rush (*Juncus balticus*), and clustered field sedge

(*Carex praegracilis*). Associated shrubs at some sites include basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*). The endemic Inyo County star-tulip (*Calochortus excavatus*) co-occurs with Owens Valley checkerbloom at some sites (Halford 1994).

Table 1. Habitat Associations for Owens Valley Checkerbloom

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Meadows and seeps	Primary	Alkaline soils; 3,580–4,650 feet elevation	CDFW 2013a; CNPS 2011

Reproduction

Owens Valley checkerbloom flowers from April through June (BLM 2011b; CNPS 2011). The pink-lavender flowers are showy and Owens Valley checkerbloom is probably an outcrossing species that is pollinated by insects. Bees are major pollinators in other related *Sidalcea* species (summarized in Leong 2006). The breeding system of Owens Valley checkerbloom is not known, but research on related *Sidalcea* species has found that several species are gynodioecious, meaning that some plants bear hermaphrodite flowers and other plants bear female-only flowers (Leong 2006). Low seed germination rates in Owens Valley checkerbloom have been reported in one study, ranging from 1.6% to 12.5% (Halford 1994). The Halford (1994) study suggested that seed weight may influence germination rates, with heavier seeds producing higher germination rates; plants may produce larger seeds in favorable years. Plant reproduction was reduced by high rates of rabbit and rodent herbivory on study sites (Halford 1994). This study identified that germination rates for Owens Valley checkerbloom may be enhanced through minor treatments such as leaching or cold stratification and mild giberellic acid treatments.

Ecological Relationships

Owens Valley checkerbloom occurs solely in mesic high-elevation alkaline meadows habitats in the Owens Valley River drainage. This species is highly restricted to a specialized habitat with very limited distribution.

The Owens Valley checkerbloom may be highly sensitive to drought conditions, although DeDecker (1978) suggested that the fleshy roots might help it survive normal drought cycles; individuals observed during the low rainfall years of 1993 and 1994 yielded low weight seeds with low viability (Halford 1994). In addition, local drought conditions may result in more browsing by rabbits and rodents, which in turn can reduce seed set and reproduction of the species (Halford 1994).

Population Status and Trends

Global: G3, Vulnerable (NatureServe 2011, Conservation Status last reviewed 2006)
State: S3, Vulnerable (CDFW 2013b)

The very restricted range and few population occurrences of Owens Valley checkerbloom make it vulnerable to declines from a variety of threats, including natural and anthropogenic sources described under Threats and Environmental Stressors. Due to the lack of long-term surveys, censuses, and/or monitoring studies, population trends of the species are unknown.

Threats and Environmental Stressors

The diversion of the Owens River and cattle grazing were the main causes of this species' decline to near extinction (DeDecker 1978). Halford (1994) reported that low annual precipitation, improper timing and intensity of cattle grazing, increased competition from rhizomatous grass species and upland shrubs, and diversions or depletions of naturally occurring water sources are all threats to the species. Lowering of the local water table by pumping and drainage for water diversion, and the resultant invasion of non-native plants, or heavy grazing and associated meadow succession may be a major threat (Hill 1993). Elmore et al. (2006), for example, reported that alkali meadow vegetation in the Owens Valley is groundwater-dependent and plant cover at groundwater-depleted sites is only weakly correlated with precipitation. Grazing, mostly by cattle, is the most frequently mentioned threat in CNDDB records (CDFW 2013a). Noxious weeds such as Russian olive (*Elaeagnus angustifolia*) and knapweed (*Centaurea* spp.) occur at a couple of occurrences, and invasion of rubber rabbitbrush (*Ericameria nauseosa*) may result from lowering of the water table.

Conservation and Management Activities

According to the CNDDB, Owens Valley checkerbloom is restricted to approximately 42 occurrences in Inyo County, of which 22 are in the Plan Area (CDFW 2013a). A cooperative project was initiated in 1994 by the BLM, the California Department of Fish and Wildlife, and The Nature Conservancy to test the long-term survivorship of reintroduced Owens Valley checkerbloom. Seeds were collected from several populations, subjected to several experimental treatments, and sown at a local nursery, and the seedlings (136 in total) were reintroduced back into sites from which the seed was collected. All plants had a minimum of a 30-centimeter (12-inch) root system when planted in October 1994, and survivorships of 50% and 85% were reported from the two sites afterwards (BLM 1994). The success of this project demonstrates that the species can be successfully propagated and transplanted, allowing some flexibility in the response of management activities to suitable habitat areas disturbed by grazing or other surface disturbing threats. However, as noted above under Threats and Environmental Stressors, groundwater management is likely a key consideration for successfully conserving and managing this species.

In 2011, the Bishop Paiute received a $200,000 grant from the USFWS to reintroduce, sustain, and nurture populations of several rare plants, including Owens Valley checkerbloom on tribal lands in the Owens Valley (USFWS 2011).

Data Characterization

An information gap extends from the mid-1990s through today. Long-term surveys, censuses, and/or monitoring studies have not been conducted on Owens Valley Checkerbloom since the mid-1990s.

Management and Monitoring Considerations

As identified under Threats and Environmental Stressors, cattle grazing, groundwater depletion, and the associated invasion by competing species are probably the main threats to Owens Valley checkerbloom. Further study regarding the response of Owens Valley

checkerbloom to these factors is needed (Halford 1994). There is no specific information available on pollinators or breeding system.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Owens Valley checkerbloom, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 147,869 acres of modeled suitable habitat for Owens Valley checkerbloom in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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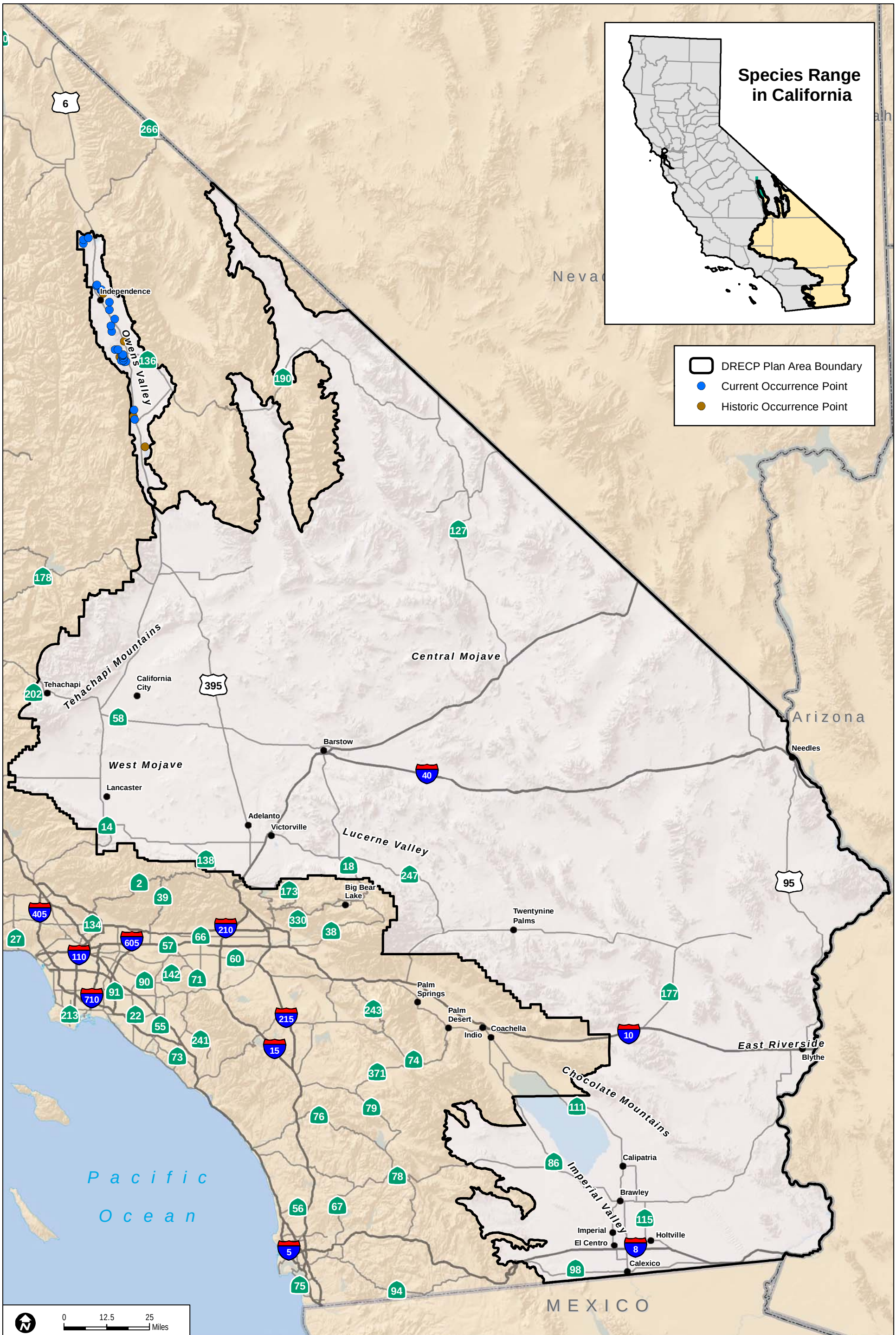

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P08

Ownes Valley Checkbloom Occurrences in the Plan Area

APPENDIX C

Species Habitat Models

APPENDIX C
Species Habitat Models

This appendix describes the species habitat modeling (also referred to as species distribution modeling) methods for the Desert Renewable Energy Conservation Plan (DRECP) and presents the species habitat model results. The description below provides an overview of the species habitat modeling method that was used to develop the habitat models for each of the proposed Covered Species. Detailed technical information on methods, data, and processing is provided at http://databasin.org/.

C.1.0 BACKGROUND

Species habitat modeling (i.e., species distribution modeling) is a necessary component of the planning process for DRECP because of the following factors:

- Need for extrapolating species and habitat distribution across areas lacking adequate data due to lack of comprehensive survey results across the Plan Area;
- Need to obtain information that will supplement existing surveys as part of the planning process;
- Need to transcend the limitations of the "snapshot in time" that survey data represents when using existing field data alone;
- Need for synthesis and analysis of multiple data sources across the entire Plan Area;
- Need to identify and rank biological values between areas; and
- Need to establish baseline conditions to compare alternate conservation strategies.

Given these factors, the DRECP Independent Science Advisors (ISA) "recommend careful use of habitat suitability models or species distribution models" (DRECP ISA 2010). Species habitat modeling can provide an objective, transparent, and repeatable means of assessing species habitat distribution where the species distribution or distribution of suitable habitat for a species is not well known. For these reasons, species habitat modeling results provide additional biological information to be used in the following components of the DRECP: conservation strategy, impact analysis, and monitoring and adaptive management. The approaches to assess the potential effects of climate change on species habitat and distribution for the DRECP are being developed and are not addressed in this document. Additionally, the approaches to address reference states for the purposes of monitoring and adaptive management for the DRECP are being developed and are not addressed in this document.

Generally, two types of models were used for the DRECP: expert-based models and statistically based models. **Expert-based models** identify species-specific habitat distribution based on scientific literature, habitat characteristics, location of documented occurrences, and expert opinion related to the physical and biological habitat parameters associated with species occurrence. As the ISA stated, expert-based models are appropriate where species occurrence data are not sufficient (i.e., too few data points to build a model) to conduct more rigorous modeling, where species occurrence data are strongly biased spatially across a plan area, or during the initial, exploratory analyses of environmental factors associated with species occurrence. **Statistically based models** specify suitable habitat and may even predict the likelihood of species occurrence based on correlations between presence/absence data and physical and biological habitat parameters. The ISA indicated that empirical, statistically based models are preferred over expert-based models (such models better control for subjective or biased input). Both expert-based models and statistically based models were developed for proposed Covered Species for the DRECP depending on species-specific considerations, including the availability of data.

The output from statistically based models is a continuous probability value ranging from 0 to 1 corresponding to range from unsuitable conditions for the species to high likelihood of species presence. The output from expert-based models is a binary result indicating suitable habitat or not. In order to use the statistically based models in conjunction with the expert-based models in developing the DRECP, a threshold value was developed for each statistically based model to convert the continuous result into a binary result.

The use of models in the DRECP conservation planning process focused on identifying areas of suitable conditions for a species (i.e., species habitat) within the Plan Area. The statistically based (i.e., Maxent) species distribution models were used in conjunction with the expert-based models to assist in the identification of potential high-priority conservation areas for the DRECP conservation strategy. Models were also used as one measure of quantification of expected conservation and effects for evaluation of conservation strategy alternatives.

C.2.0 SPECIES HABITAT MODEL DEVELOPMENT

Species habitat models have been developed for the 37 proposed Covered Species under the DRECP. The following summarizes the process for developing the DRECP species habitat models.

Early in the DRECP planning process, existing published species distribution models for proposed Covered Species were gathered and evaluated. Additionally, early versions of expert-based and Maxent models were developed for the DRECP. These early model versions were used to support the initial DRECP planning process and were

documented in previous versions of the draft Baseline Biology Report (Dudek and ICF 2012) and the Description and Comparative Evaluation of the Draft DRECP Alternatives (DRECP REAT 2012).

In order to continue to refine and improve the species habitat models, the models documented in Dudek and ICF 2012 went through the following review process:

1. Outside Expert Review (Winter–Spring 2012). This involved the individual review of species profiles and species habitat models by outside scientists and species experts. Comments on profiles have been integrated in the profiles in Appendix B of this document. Comments on species habitat models were used to refine the species habitat models.
2. Independent Science Panel Review (Summer 2012). This involved a panel review of the science used in the DRECP. Comments on species habitat models were used to refine the species habitat models.
3. DRECP Species Modeling Forum (January 2013). Researchers and modelers with expertise in species distribution modeling were gathered with REAT agency biologists to review existing species habitat models and provide species-by-species recommendations on data sources and modeling approaches, as well as address issues common to species modeling in general (including technical issues, such as thresholds, raised in DRECP independent science reviews). For taxa with multiple available models, this forum allowed selection of the one most relevant to the DRECP's purposes and discussion of the differences among the various models for a given taxon. Experts from the Conservation Biology Institute (CBI), University of California Berkeley (UCB), University of California Davis (UCD), University of California Santa Barbara (UCSB), and the United States Geological Survey (USGS) collaborated to develop the recommendations. These scientists also provided recommendations and advice on specific technical issues arising during the DRECP species model development work but subsequent to the forum.

This comprehensive input gathering process provided robust input from species experts, agency specialists, and modelers, and was used to scientifically vet, refine, and improve the DRECP species habitat models for all proposed Covered Species. Statistically based Maxent models were used for a majority of the DRECP Covered Species. Where statistically based models were not recommended due to data limitation or species-specific considerations, expert-based models were developed. Species habitat models used for DRECP were developed by several entities, including CBI, Dudek, UCB, UCD, UCSB, and USGS.

The model results for each species are provided in this appendix. Supporting documentation with detailed information on methods, data, and processing is provided on http://databasin.org/.

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Parish's Daisy (*Erigeron parishii*)

Photo courtesy of Duncan S. Bell.

Legal Status

State: S2S3[1]
CNPS: Rare Plant Rank 1B.1[2]
Federal: Threatened
Critical Habitat: Originally designated on December 12, 2002 (67 FR 78570–78610).
Recovery Planning: *San Bernardino Mountains Carbonate Plants Draft Recovery Plan* (USFWS 1997)
Notes: No status changes predicted by U.S. Fish and Wildlife Service (USFWS) in 2010 (75 FR 28636–28642)

Taxonomy

Parish's daisy *(Erigeron parishii)* was named by Asa Gray in 1884 in his *Synoptical Flora of North America* and has remained stable with no changes since. Parish's daisy is in the sunflower family (Asteraceae) (IPNI 2011). It is an herbaceous perennial subshrub approximately 7 to 30 centimeters (3 to 12 inches) in height from its taproot. A full physical description of the species can be found in *Jepson eFlora* (Jepson Flora Project 2012).

Distribution

General

Parish's daisy is endemic to Southern California, restricted to dry, calcareous (mostly limestone) slopes of the San Bernardino Mountains, with a few collections from granitic areas at the east end of the San Bernardino Mountains and in the Little San Bernardino Mountains (Neel 2000; Sanders 2006). Parish's daisy occurs at elevations between 3,700 and 6,600 feet, most often in washes and canyon bottoms, but sometimes on alluvial benches or steep rocky

[1] **S2:** Imperiled.
[2] **1B:** Rare, threatened, or endangered in California and elsewhere; **X.1:** Seriously endangered in California.

mountainsides (Mistretta and White 2001). It is estimated that 1,029 acres are occupied Parish's daisy habitat (USFWS 2009).

Distribution and Occurrences within the Plan Area

Historical

Parish's daisy was first described by Asa Gray in 1884 from specimens collected by S.B. Parish at Cushenbury Springs in May 1881 (Abrams and Ferris 1960; Krantz 1979). It was reported to be "abundant on stony hillsides at Cushenberry Springs" by Hall (1907), although it is unclear whether Hall was referring to Parish's collections of the species (Sanders 2006). Within the Plan Area, the California Natural Diversity Database (CNDDB) includes two historical occurrences that were documented in 1988 and two historical occurrences for which status is unknown (Figure SP-P16). However, each of these occurrences is presumed to be extant.

Recent

Within the Plan Area, the CNDDB includes 40 recent occurrences (i.e., post-1990) of Parish's daisy and all are regarded as extant (CDFW 2013a; Figure SP-P16). The populations occur primarily on U.S. Forest Service (USFS) and BLM lands, but two of the populations on USFS and BLM lands also extend onto private lands within the Plan Area. Two populations occur within the Joshua Tree National Park and another is located on the University of California Natural Reserve System Burns Pinion Ridge Reserve (CDFW 2013a).

In 2009 the USFWS determined that the range and distribution of this species was essentially the same as it was at the time of listing (1994).

Natural History

Habitat Requirements

Parish's daisy occurs in Mojavean desert scrub and pinyon and juniper woodlands (CNPS 2011) and is largely restricted to loose, carbonate alluvium, although it is occasionally found on other rock types (Sanders 2006) (Table 1). Populations of Parish's daisy are most commonly found along washes on canyon bottoms or on loose alluvial

deposits on adjacent benches, but they are also occasionally found on steep rocky slopes (Sanders 2006). Based on this species' occurrence on noncarbonate granitic soils, it is possible that the apparent carbonate preference is due to reduced competition from other plants, although reports of this species on noncarbonate soils are few (Sanders 2006). It has also been observed at sites where soils have been found to be strongly alkaline, implying that the noncarbonate granitic soils may have been influenced in their soil chemistry by adjacent carbonate slopes (Sanders 2006).

Specific plant species associated with Parish's daisy have not been described in the literature, but dominant species within pinyon and juniper woodland where Parish's daisy is typically found include singleleaf pinyon pine (*Pinus monophylla*), Utah juniper (*Juniperus osteosperma*), and more rarely California juniper (*Juniperus californica*) and western juniper (*Juniperus occidentalis*). Understory species within pinyon and juniper woodland are more variable, but may include mountain-mahogany (*Cercocarpus ledifolius*), Mormon tea (*Ephedra viridis*), Mojave yucca (*Yucca schidigera*), Joshua tree (*Yucca brevifolia*), and encelia (*Encelia* sp.).

Parish's daisy co-occurs with another carbonate endemic, Cushenbury oxytheca (*Acanthoscyphus parishii* var. *goodmaniana*). Its presence, however, appears to be negatively related to at least two other carbonate soils species—Cushenbury milk-vetch (*Astragalus albens*) and Cushenbury buckwheat (*Eriogonum ovalifolium* var. *vineum*)—which tend to occur on more stable slopes.

Table1. Habitat Associations for Parish's Daisy

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Pinyon-juniper woodland, Joshua tree woodland, Mojavean desert scrub, Jeffrey pine-western juniper woodland	Primary habitat	Carbonate soils (limestone), 3,000 to 6,600 feet	Sanders 2006; USFWS 2009

Reproduction

Parish's daisy is a long-lived perennial (Mistretta and White 2001) that flowers from May through August (CNPS 2011), peaking mid-May to mid-June (Sanders 2006). Based on the conspicuous flowers, pollinators are probably insects and would include bees, butterflies, and other known pollinators of similar and related species (Sanders 2006). Parish's daisy produces plumed achenes adapted for wind dispersal (Mistretta and White 2001) and does not appear to have a seed dormancy mechanism (Mistretta 1994). Based on observations of seedlings at several sites (Krantz 1979), reproduction is probably primarily by seed rather than vegetatively by rhizomes or stolons. A recent study by Neel and Ellstrand (2001) found no evidence of vegetative reproduction, concluding that the species probably primarily reproduces sexually through outcrossing.

Recent research on allozyme diversity showed that genetic diversity was high (compared to many narrowly endemic plant taxa) and populations were only moderately differentiated, suggesting that gene flow among populations is still high and any recent fragmentation has not yet affected genetic diversity. Maintaining the existing large population sizes is an important component in maintaining gene flow among populations (Neel and Ellstrand 2001).

Population Status and Trends

Global: G2, Imperiled (NatureServe 2011, Conservation Status last reviewed 2006)
State: S2, Imperiled (CDFW 2013b)

The current population status of Parish's daisy is unclear and there is a discrepancy in total reported occurrences of the species. According to the final listing rule in 1994, Parish's daisy was known from fewer than 25 occurrences with a total estimated population size of 16,000 individuals, but at that time, the San Bernardino National Forest had mapped 87 site-specific occurrences (USFWS 2009). USFWS (2009) notes that what constitutes an occurrence has been subjectively defined over various surveys, making it difficult to specify status or change in status of Parish's daisy since it was listed. In addition, there has been an increase in survey efforts for this species since listing that

has resulted in an increase in the number of occurrences detected. Sanders (2006) characterizes Parish's daisy as one of the more common carbonate endemics of the San Bernardino Mountains. Nonetheless, there have not been any systematic population studies conducted over time to document population trends.

Threats and Environmental Stressors

The main threat to Parish's daisy is limestone mining because this species is mostly restricted to carbonate deposits (USFWS 2009). Besides direct impacts, dust and artificial lighting can affect the species through dust impacts on soil chemistry and lighting availability for seeds and the impacts of artificial lighting on growing conditions (USFWS 2009). Sanders (2006) notes that after moistening, the mining dust appears to harden into a cement-like coating. Additional threats listed by USFWS and CNPS include energy development projects, off-highway vehicles, fuel-wood collection, fire suppression activities, camping, target shooting, road construction, and residential developments, but these threats are relatively low compared to mining (USFWS 2009; CNPS 2011).

The specific potential effects of climate change on Parish's daisy are unknown, but if climate change caused a shift to higher elevations due to warmer and drier conditions, as has occurred with other plant species on the Santa Rosa Mountains of Southern California (Kelley and Goulden 2008), this endemic species could be concentrated in a smaller area and more vulnerable to extinction (USFWS 2009).

Conservation and Management Activities

The *San Bernardino Mountains Carbonate Plants Draft Recovery Plan*, prepared by the USFWS in 1997, addressed Parish's daisy and four other federally listed species: Cushenbury buckwheat, Cushenbury milk-vetch, San Bernardino Mountains bladderpod (*Physaria kingii* ssp. *bernardina*), and Cushenbury oxytheca (USFWS 1997). The Recovery Plan for these species included the following recovery criteria:

1. Sufficient habitat protected in a reserve system for persistence of existing populations in their ecological context, including the largest populations and best and manageable habitat

2. Identification of potential buffer zones, although not necessarily secured, with an estimate of 4,600 acres needed for habitat connectivity, buffers, and a natural community context
3. Population monitoring and habitat management to provide for early detection of population instability in the reserve system
4. Expansion of existing populations or reintroductions to reduce the chance of extinction due to randomly occurring events.

Based on these recovery criteria, the Recovery Plan identified the following actions:

1. Protect significant extant populations in a reserve system on federally owned land, which would include buffer zones, and maintain selection habitat connections
2. Restore habitat and conduct reintroductions and/or population enhancements where appropriate and feasible
3. Identify and implement appropriate management measures
4. Monitor populations
5. Conduct limited surveys and taxonomic assessments to find new populations.

The Recovery Plan identified the USFS, BLM, California Department of Fish and Game, and USFWS as the agencies primarily involved in the recovery effort (USFWS 1997).

In 2003, the *Carbonate Habitat Management Strategy* (CHMS) was developed by the USFS and BLM in collaboration with a Working Group consisting of mining interests, private landowners, and conservation groups to address impacts to the five federally listed plants associated with carbonate habitats (Olsen 2003). The CHMS, which covers about 160,000 acres (called the Carbonate Habitat Management Area or CHMA), has three main objectives:

1. Economic: regulatory certainty for mining activities, protection of the viability of mining, and streamlining and cost reduction of the permitting process
2. Conservation: maintenance and management of geomorphic and ecological processes of the landscape and placement of

habitat blocks to maintain the carbonate plants, to avoid jeopardy (per Section 7 of the federal Endangered Species Act) and adverse modification or destruction of critical habitat, to contribute to recovery, and to avoid future listings

3. Regulatory: streamlining of permitting, California Environmental Quality Act (CEQA) review, streamlining of County implementation of the California Surface Mining Reclamation Act, and to allow BLM and USFS to comply with certain court-ordered stipulations stemming from lawsuits (i.e., *Center for Biological Diversity v. BLM* and *Southwest Center for Biological Diversity v. Sprague*).

The CHMS includes delineation of an Initial Habitat Reserve, designation of Conservation Units within the CHMA whereby loss and conservation of habitat values can be objectively measured, and contribution by federal agencies and mining interests to reserve assembly through various mechanisms (e.g., dedication of existing unclaimed federal land, purchase of private lands or lands with mining claims, land exchanges, or conservation banking) (Olsen 2003).

Upon successful completion, the CHMS would meet or exceed recovery criteria 1 and 2 listed previously (USFWS 2009).

Implementation of the CHMS has been incorporated by the USFS into the Land Management Plans for the Angeles and San Bernardino National Forests (USFS 2005) and by the BLM into the West Mojave Plan (BLM 2005).

Within the Plan Area, a large percentage of the known populations occur on BLM-administered lands that are covered under the West Mojave Plan (BLM 2005). However, it is estimated by the USFWS that 73% of these lands are under claim to mining companies and development of these sites will make conservation difficult (Sanders 2006). One population around Three Sisters Peak West is under non-profit control, which presumably will have conservation benefits for the species.

Data Characterization

The general distribution of Parish's daisy is fairly well known, based on its close association with carbonate substrates and increased

survey efforts since its federal listing as endangered in 1994 (67 FR 78570–78610). However, its population status in terms of population trends is not well understood due to subjective mapping of occurrences between the different survey efforts and a lack of systematic studies carried out over time (USFWS 2009).

Management and Monitoring Considerations

To achieve species recovery, the USFWS (2009) has identified several management and monitoring strategies that need to be implemented for Parish's daisy. These strategies include:

1. Working with the San Bernardino National Forest to conduct systematic monitoring of Parish's daisy throughout known and potentially occupied sites
2. Within occupied Parish's daisy habitat continue monitoring programs for the effectiveness of measures to protect the species from recreation activities
3. Avoid new developments in or near Parish's daisy habitat.

Research by Mistretta and White (2001) indicates that restoration of Parish's daisy population can be successful. A total of 66% of plants transplanted to a disturbed but irrigated site in 1991–1992 survived a 6-year monitoring period. In addition, successful recruitment of progeny was reported at the restoration site. Sanders (2006) suggests that Parish's daisy may be better able to recover after disturbance than some carbonate endemics.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Parish's daisy, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 187,517 acres of modeled suitable habitat for Parish's daisy in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat for Parish's daisy in the Plan Area.

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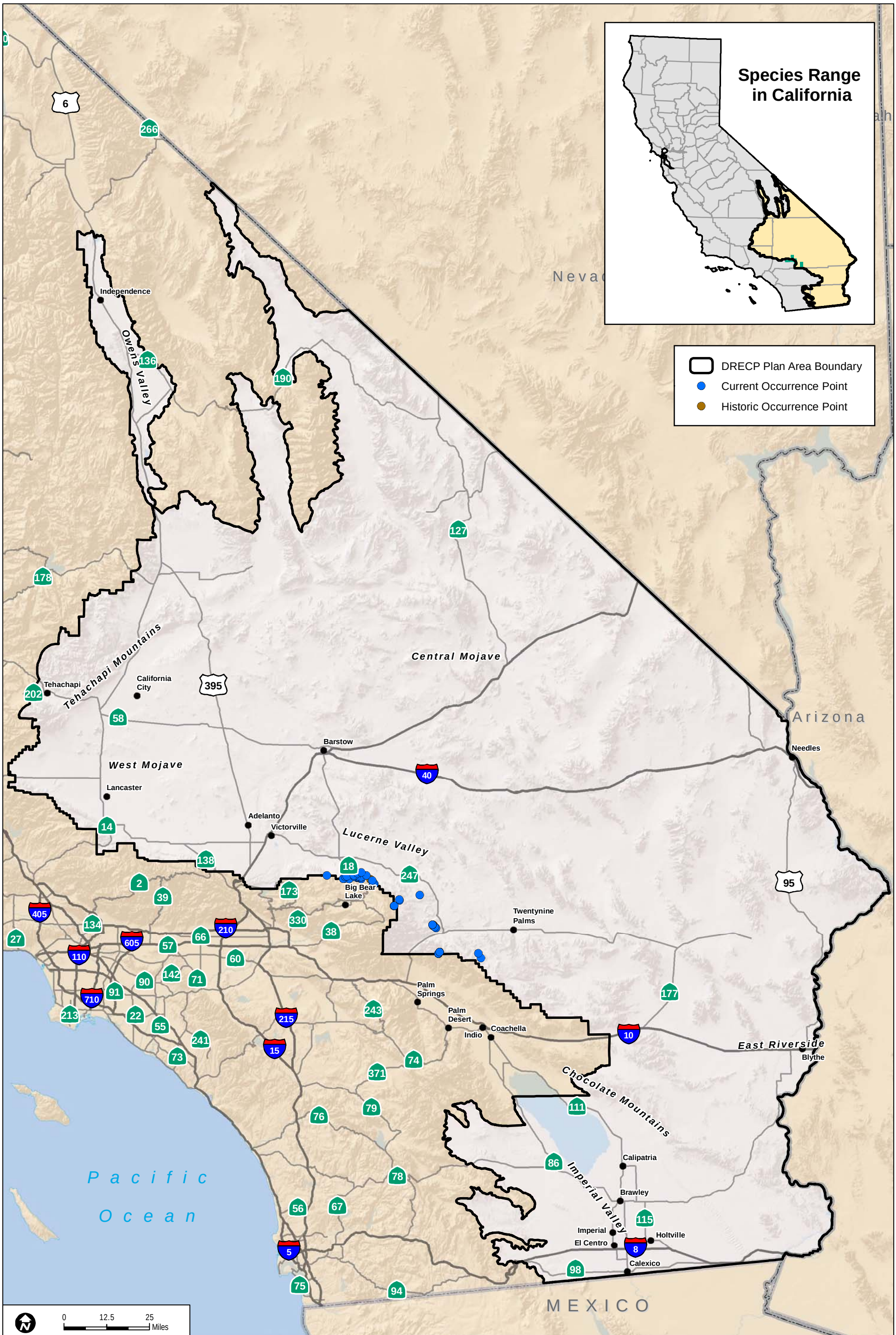

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P09

Parish's Daisy Occurrences in the Plan Area

Triple-Ribbed Milk-Vetch (*Astragalus tricarinatus*)

Photo courtesy of John Green.

Legal Status

State: S1.2[1]
California Rare Plant Rank: 1B.2[2]
Federal: Endangered; U.S. Forest Service Sensitive
Critical Habitat: N/A
Recovery Planning: N/A
Notes: The federal 5-year review of the species recommended no change needed for the endangered status of the species (USFWS 2009).

Taxonomy

Triple-ribbed milk-vetch (*Astragalus tricarinatus*) was first described by Asa Gray in 1876, based on a collection from Whitewater Canyon (63 FR 53596–53615). Although it was transferred to another genus—*Hamosa*—in 1927, this species is currently accepted as *Astragalus tricarinatus* and there is no available information to suggest that the taxonomy of triple-ribbed milk-vetch is uncertain or in question (Jepson Flora Project 2011).

Triple-ribbed milk-vetch is a short-lived, perennial herb with stems approximately 5 to 25 centimeters (2 to 10 inches) in length. A full physical description of the species can be found in the *Jepson eFlora* (Jepson Flora Project 2011).

Distribution

General

The general range of triple-ribbed milk-vetch includes the eastern San Bernardino Mountains/Whitewater Canyon area, Morongo Canyon,

[1] **S1:** Critically imperiled; **X.2:** Threatened.
[2] **1B:** Rare, threatened, or endangered in California and elsewhere; **X.2:** Fairly endangered in California.

and the western part of the Little San Bernardino Mountains, with disjunct occurrences in the Orocopia (Barneby 1959) and Santa Rosa mountain ranges (Figure SP-P18), although the Orocopia occurrence is unvouchered (USFWS 2009). Throughout the species' range, there are 21 occurrences, of which, 19 are considered extant (CDFW 2013a). Within the Plan Area, triple-ribbed milk-vetch occurs in the Morongo Canyon area and in the Little San Bernardino Mountains at Coyote Hole Spring, Long Canyon, and possibly at Keys Ranch.

Distribution and Occurrences within the Plan Area

Historical

Historically (prior to 1990), triple-ribbed milk-vetch was known from Whitewater and Morongo canyons in Riverside and San Bernardino counties and southeast to the Orocopia Mountains in Riverside County (63 FR 53596–53615). The California Natural Diversity Database (CNDDB) includes no historical occurrences in the Plan Area (CDFW 2013a). A 1926 collection from a small population is also noted from Coyote Hole Spring along the northern edge of the Little San Bernardino Mountains and south of the town of Joshua Tree (USFWS 2009), but no recent information is available for this site, and the occurrence is not in the CNDDB (CDFW 2013a). The Keys Ranch site in Joshua Tree National Park is also from 1926 but it was not detected in a 1999 survey (USFWS 2009).

Recent

This description of recent occurrences is primarily taken from the 2009 5-year review of triple-ribbed milk-vetch (USFWS 2009) because it includes all of the CNDDB occurrences in the Plan Area as well as some occurrences that are not in the CNDDB. As shown in Figure SP-P18, there are 21 recent occurrence locations for triple-ribbed milk-vetch in the Plan Area: Wathier Landing, Catclaw Flat, Mission Creek, Dry Morongo Canyon and Wash, Big Morongo Canyon, Long Canyon, Coyote Hole Spring, Key's Ranch (note that this site is unvouchered), and Orocopia Mountains. The characterization of the species' distribution is complicated by the fact that the occurrences appear to represent different types of populations: source populations, waifs (i.e., isolated plants), and deme populations (i.e., groups of isolated plants) (USFWS

2009). Source populations are larger, permanent populations (i.e., up to several hundred individuals) typically located in the upper watershed areas. Waifs are scattered individuals in washes downstream of source populations. Deme populations are discrete or isolated groups of waifs that may exhibit intra-population breeding but do not persist. Habitats associated with these population types are discussed in more detail in Habitat Requirements.

There are two recognized source populations in the Plan Area: Wathier Landing and Catclaw Flat. The Wathier Landing population, which is in the Mission Creek drainage just east of Wathier Landing, supported at least 300 aboveground individuals in 2004 (White 2004) and more than 300 adult individuals and many seedlings in 2005 (Amsberry and Meinke 2007). The Catclaw Flat occurrence was first discovered in 2005 about 2.5 miles from the Wathier Landing site and consisted of about 100 individuals, including seedlings (Amsberry and Meinke 2007). Both sites are conserved on private land owned by The Wildlands Conservancy (TWC).

The other occurrences in the Plan Area are considered deme populations that are not self-sustaining (USFWS 2009). Besides the Wathier Landing and Catclaw Flat source populations, the largest documented population was in Big Morongo Canyon; this population numbered less than 50 individuals in 1993, but a survey of the site in 2005 failed to detect the species (CDFW 2013a). One large reproductive individual (but no seedlings) was found in 2005 on a slide of exposed, decomposed granite on the canyon wall in Big Morongo Canyon (Amsberry and Meinke 2007) within the Bureau of Land Management (BLM) Big Morongo Canyon Reserve (CDFW 2013a). Two waif individuals were detected in Long Canyon in Joshua Tree National Park in 2006 (CDFW 2013a).

It should be noted that botanists suspect that more populations of triple-ridged milk-vetch exist on upland slopes in suitable habitat (e.g., rocky, exposed slopes and ridges), but the rugged terrain occupied by this species makes exploration difficult, and small plants tend to blend in with light-colored granitic substrates, making them hard to detect (Amsberry and Meinke 2007).

Natural History

Habitat Requirements

Triple-ribbed milk-vetch is characterized as generally occurring in Joshua tree woodland and Sonoran desert scrub (see Table 1) (CDFW 2013a; CNPS 2011). Throughout its range, it occurs at elevations of 1,300 to 4,000 feet above mean sea level (amsl) (USFWS 2009). Occurrences within the Plan Area occur at 2,300 to 3,700 feet amsl. However, as discussed in Recent Occurrences, populations are characterized as source populations, deme populations, and waifs. The focus of this description is habitat for source populations because they are considered the most important element for the species for conservation purposes. The deme populations and especially the waif populations that likely occur from seedlings washed downstream and downslope from source population are small and not self-sustaining and, therefore, are not as important for conservation and management. These sites are not the primary habitat for the species (Amsberry and Meinke 2007), and these small ephemeral populations likely do not contribute to long-term viability of the species. However, waifs in the Whitewater Canyon wash area are on an eroded talus of the same soil type that occurs in primary habitat for the source populations (Barrows, pers. comm., 2012).

Table1. Habitat Associations for Triple-Ribbed Milk-Vetch

Land Cover Type	Habitat Designation	Habitat Parameters	Supporting Information
Mojave mixed woody scrub, Sonoran desert scrub	Primary habitat for source populations	Granitic substrates Elevation 1,300 to 4,000 feet amsl	White 2004 Amsberry and Meinke 2007 CDFW 2013a; USFWS 2009

The Wathier Landing source population occurs on an outcrop of metamorphic rock which is weathering into “unproductive-looking” gravelly soil at about 3,700 feet amsl (White 2004). Triple-ribbed milk-vetch was not detected in surrounding granitic slopes or alluvial fans and washes (White 2004). The substrate where the plants were

actually detected was largely bare of other species, but associated plants included giant needlegrass (*Achnatherum coronatum*), California buckwheat (*Eriogonum fasciculatum*), desert ceanothus (*Ceanothus greggii*), tree poppy (*Dendromecon rigida*), bigberry manzanita (*Arctostaphylos glauca*), bitter snakewood (*Condalia globosa*), hairy yerba santa (*Eriodictyon trichocalyx*), and Mojave yucca (*Yucca schidigera*) (Amsberry and Meinke 2007; White 2004). The Catclaw Flat population was located on decomposed granite substrate on an exposed ridge at about 3,400 feet amsl in association with the same plant species as the Wathier Landing site (Amsberry and Meinke 2007).

The unique soil association is a critical component of the species distribution, although the mechanism for that association is unclear. Little else grows on these soils, but whether it is the lack of competition, a unique chemical composition, or the appropriate level of erosion-disturbance that has fostered the plant soil association has yet to be understood. Where that soil occurs, or where similar soil outcrops occur, triple-ribbed milkvetch is often found. In Mission Creek, on these soil types, but in relatively flat terrain, this milkvetch has been observed primarily after a large disturbance (wildfire with firefighting related soil disturbance) (Barrows, pers. comm. 2012).

Triple ribbed milk-vetch generally occurs in dry washes, at the bases of canyon slopes, and on steep scree slopes (USFWS 2009). Generally, primary habitat for source populations in the Plan Area consists of rocky slopes and ridges that are mostly barren. Notably the two source populations are at the two highest elevations of all of the occurrences in the Plan Area, supporting the notion that the large source populations occur in upslope areas in the upper watersheds and the smaller deme populations and waifs occur at lower elevations in downstream washes and downslope (White 2004; USFWS 2009).

Reproduction

Triple-ribbed milk-vetch is a short-lived, perennial member of the pea family (USFWS 2009). Some species-specific life history information is available for this species and comes from a single study of the species conducted in 2005 and 2006 by Amsberry and Meinke (2007) at the two source populations in Wathier Landing and Catclaw Flat.

The blooming season for triple-ribbed milk-vetch is February through May (CNPS 2011). Amsberry and Meinke (2007) found that 62% of sample individuals at Wathier Landing were in flower in March 2005, and 38% were beginning to produce fruit. At Catclaw Flat, all sampled plants were in fruit in May 2005. Sampled plants at Catclaw Flat reproduced an estimated mean of 2,759 seeds per plant, which is higher than reported rates for other members of this genus. Hundreds of seedlings were observed at both sites in 2005, which was a high rainfall year (a "good" rainfall year), and seedlings were also observed in 2006, which was a dry year. White (2004) also observed seedlings at the Wathier site in 2004, suggesting that reproduction and seedling germination may occur in most years at these source populations (Amsberry and Meinke 2007).

In a pilot greenhouse study of germination requirements of triple-ribbed milk-vetch, Amsberry and Meinke (2007) found that 80% of "viable-appearing" seeds germinated within 72 hours after scarification and wetting; scarification probably occurs naturally through exposure and/or the action of tumbling gravel during flooding. Amsberry and Meinke (2007) also found that growth was more robust in pots inoculated with soil from vigorous, cultivated plants of the obligately mycorrhizal species *Astragalus applegatei* that were previously inoculated with native soil containing mycorrhizae and Rhizobium.

Despite the apparent high productivity of this species, the 5-year review for the species states that "the abundance of this species fluctuates from year to year and may not be present above ground in drought years" (USFWS 2009, p. 1). Long-term studies of this species have not been conducted to determine its response to wet and dry cycles.

Amsberry and Meinke (2007) noted that all mature reproductive individuals appeared to be perennial and many had obvious woody bases. The longevity of individuals is suspected to be 3 to 5 years, but long-term studies are needed (Amsberry and Meinke 2007).

Pollinators of triple-ribbed milk-vetch are unknown. Amsberry and Meinke (2007) noted that field conditions were too windy to observe

pollinators but indicate that the species' showy flowers are typical of legumes pollinated by native bees and honeybees.

Dispersal mechanisms are unknown, but observations of many seedlings around mature reproductive plants suggest that dispersal occurs over short distances within the source populations (Amsberry and Meinke 2007; White 2004). The deme populations and waifs probably stem from seeds washed downstream or downslope from the source populations (USFWS 2009; White 2004).

Ecological Relationships

Little is known about the ecological relationships of triple-ribbed milk-vetch. The 5-year review for the species indicates that the individuals may not appear aboveground during drought years (USFWS 2009), but Amsberry and Meinke (2007) suggest that reproduction and seedling germination may occur in most years at the source populations. Long-term studies are needed to understand the species' response to wet and dry cycles.

The pilot greenhouse study by Amsberry and Meinke (2007) found a positive growth response in soils from the obligately mycorrhizal congener *Astragalus applegatei*, raising the potential importance of relationships with fungal or bacterial associates.

Pollination and dispersal studies have not been conducted, although the species' showy flowers may attract native bees and honeybees, and seedlings are readily observed around source populations (Amsberry and Meinke 2007; White 2004).

Associated plants at the two source populations in the Plan Area—Wathier Landing and Catclaw Flat—are similar, but this similarity is not unexpected because of the close proximity of the two sites. The plant communities at most other occurrences have not been described, but the vegetation community at the East Deception Creek site, which is a deme population of about 50 individuals on a scree slope, includes creosote bush (*Larrea tridentata*), Schott's indigobush (*Psorothamnus schottii*), rush milkweed (*Asclepias subulata*), burrobush (*Ambrosia salsola* var. *pentalepis*, and deerweed (*Acmispon glaber*) (Le Doux 2007, cited in USFWS 2009).

Given that most occurrences of triple-ribbed milk-vetch are in barren areas, local plant associations do not appear to be an important factor for presence or absence.

Population Status and Trends

Global: G1, Critically Imperiled (NatureServe 2011, Conservation Status last reviewed 2003)
State: S1, Critically Imperiled (CDFW 2013b)

Other than the site-specific counts and population estimates for the approximately 18 extant occurrences for triple-ribbed milk-vetch, there are little data for population status and trends. For the 5-year review of the species, the U.S. Fish and Wildlife Service (USFWS) estimated the known rangewide population to be less than 500 individuals, including source and deme populations and waifs (USFWS 2009). The two observed source populations in the Plan Area—Wathier Landing and Catclaw Flat—were known to support approximately 300 and 500 individuals, respectively, in the mid-2000s (Amsberry and Meinke 2007), but their current status is unknown. The other occurrences in the Plan Area are small, unsustainable deme populations and waifs (see Recent Occurrences). However, the actual population is likely to be substantially larger because not all suitable habitat areas have been surveyed. The observed deme populations and waifs in downstream and downslope areas indicate the likely presence of larger, but as yet unknown, upslope source populations (USFWS 2009).

Threats and Environmental Stressors

The main anthropogenic threats to triple-ribbed milk-vetch that triggered the federal listing of the species in 1998 was bulldozing for maintenance of a gas pipeline and earth-moving activities along a stretch of Big Morongo Canyon to realign segments of a crude oil pipeline that had been exposed during winter storms in 1992–1993 (63 FR 53596–53615). It is considered to be under continuing threat from maintenance of the crude oil pipeline and from off-highway vehicle use in the canyons. Its small population numbers make it vulnerable to stochastic events and anthropogenic events such as pipeline leaks (USFWS 2009). New threats identified since the species' federal listing include wildland fire suppression activities, flooding,

and climate change (USFWS 2009). Amsberry and Meinke (2007) also identify exotic weed infestations resulting from increased vehicle and foot traffic as a potential threat to the species.

Rangewide, but outside the Plan Area, other potential threats include residential development of population location in East Deception Canyon and Lower Mission Creek, which may affect downstream habitat and facilitate off-highway vehicle use (USFWS 2009).

Conservation and Management Activities

Conservation and management activities within the Plan Area include preservation of the two known source populations—Wathier Landing and Catclaw Flat—on the Whitewater Preserve, privately owned by TWC. These lands are operated and managed with the same goals as the surrounding BLM San Gorgonio Wilderness Area (USFWS 2009). TWC also leased a nearby 40,032-acre BLM grazing allotment that has since been relinquished, and grazing is no longer permitted (USFWS 2009).

Small populations of triple-ribbed milk-vetch occur in Big Morongo Canyon in the Plan Area within the BLM Big Morongo Canyon Preserve, which is designated an Area of Critical Environmental Concern (ACEC), encompassing about 31,000 acres. Further, the San Gorgonio Additions Wilderness Area comprises approximately 39,215 acres between San Bernardino National Forest and the Morongo Valley; it includes significant portions of the Mission Creek and Whitewater drainages, and preserves significant contiguous occurrences and contiguous habitat (USFWS 2009). It is highly possible that additional source populations within the Plan Area occur in the San Gorgonio Additions Wilderness Area, given the nearby locations of the Wathier Landing and Catclaw Flat source populations (see Figure SP-P18).

The Long Canyon and Keys Ranch occurrences are within Joshua Tree National Park, and as of 2009 a management plan was being prepared for the species (USFWS 2009).

Conservation of the species outside the Plan Area is provided by the Coachella Valley Multiple Species Habitat Conservation Plan

(MSHCP), which conserves 2,838 of the 3,007 acres of modeled habitat distributed across Whitewater Canyon (1,295 acres), Mission Creek and Big Morongo Canyon (819 acres), Whitewater floodplain (866 acres), and Santa Rosa and San Jacinto Mountains (1 acre) (CVMSHCP 2007).

Data Characterization

The geographic range of triple-ribbed milk-vetch probably is fairly well known since no new outlier populations have been discovered since 1985 (the Agua Alta site in the Santa Rosa Mountains). The Orocopia Mountains occurrence is unvouchered. However, within the species' geographic range boundaries, its distribution probably is still not well understood. Only two source populations for the species that are in close proximity to each other have been documented—the Wathier Landing and Catclaw Flat occurrences on TWC land. The other documented occurrences are deme populations and waifs that indicate a larger upslope source population that has not been documented but provides seedlings for the downstream and downslope populations (USFWS 2009). Because of the rugged and potentially inaccessible primary habitat for the species (i.e., rocky slopes, canyon walls, and ridges in remote upper watershed areas), much suitable habitat probably has not been adequate surveyed. In addition, if the species' abundance and detectability varies in relation to wet and drought cycles, it may not be detectable on occupied sites in a dry year and follow-up surveys would be required (USFWS 2009). Also, smaller individuals are difficult to detect from a distance because they blend in with the light-colored granitic substrates on which they occur (Amsberry and Meinke 2007). For these reasons, it is likely that the current distribution information significantly underestimates its actual distribution.

Management and Monitoring Considerations

The 5-year review for triple-ribbed milk-vetch (USFWS 2009) recommended several actions related to management and monitoring of the species, including:

- Demographic and survival studies at known sites

- Predictive habitat modeling involving source soils to locate new source populations
- Site-specific fire suppression plans, including avoidance areas, bulldozer lines, and aerial retardant drops, as well as post-fire surveys
- Development of protocols to ensure low impacts during facilities maintenance (e.g., pipelines).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for triple-ribbed milk-vetch, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 81,251 acres of modeled suitable habitat for triple-ribbed milk-vetch in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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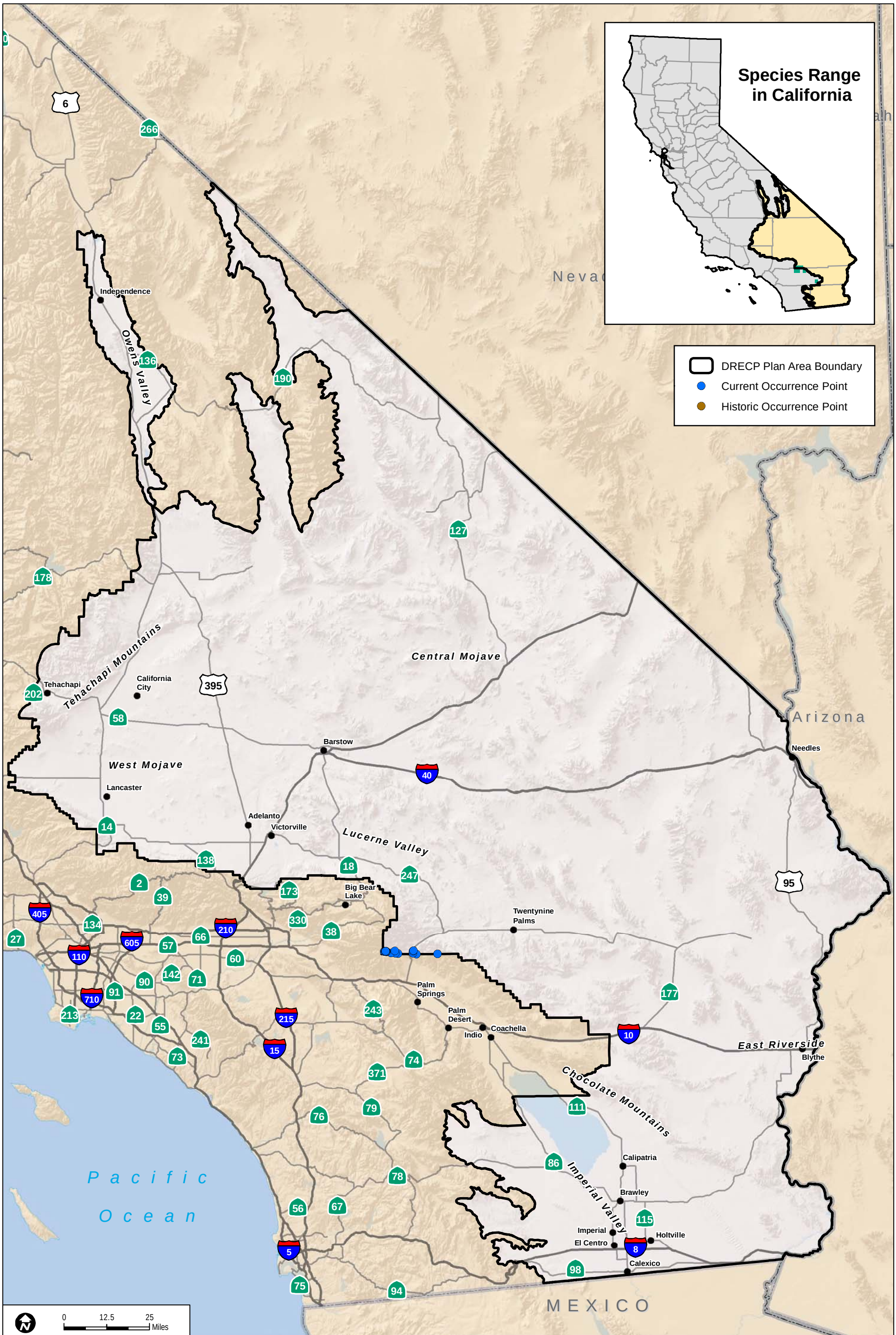

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-P10

Tripple-ribbed Milk-vetch Occurrences in the Plan Area

Western Yellow-Billed Cuckoo (*Coccyzus americanus occidentalis*)

Courtesy of Murrelet Halterman, PhD.

Legal Status

State: Endangered
Federal: Candidate, Bureau of Land Management Sensitive, U.S. Fish and Wildlife Service Bird of Conservation Concern, U.S. Forest Service Sensitive
Critical Habitat: N/A
Recovery Planning: N/A
Notes: In 2001, the U.S. Fish and Wildlife Service (USFWS) completed a 12-month review of a petition for listing the western yellow-billed cuckoo (*Coccyzus americanus occidentalis*) under the federal Endangered Species Act, and it determined that a listing was warranted but precluded at the time by higher priority listing actions, at which time the subspecies was added to the candidate list (66 FR 38611–38626). The most annual recent review of candidate species by the USFWS on October 26, 2011 includes the species yellow-billed cuckoo (*Coccyzus americanus*), with a western U.S. Distinct Population Segment (DPS) (76 FR 66370-66439) (i.e., the review does not refer to the western yellow-billed cuckoo subspecies even though California lists the subspecies *C. a. occidentalis* as Endangered). The USFWS continues to find that the western U.S. DPS warrants listing, but that such listing was precluded at the time of 2011 review. The USFWS states that they are working on a proposed listing rule that they expect to publish before making the next annual resubmitted petition 12-month finding (76 FR 66370-66439).

Taxonomy

Two subspecies of the yellow-billed cuckoo are recognized—western yellow-billed cuckoo (*C. a. occidentalis*) and eastern yellow-billed cuckoo (*C. a. americanus*)—although the validity of the taxonomic grouping has been debated based on morphometric measurements

(e.g., wing length) (Banks 1988, 1990; Franzreb and Laymon 1993). Banks (1988) initially found statistically insignificant differences in wing length, bill length, and upper mandible depth between alleged subspecies. Revised analyses were performed given statistical and methodological errors in the Banks (1988) study. The updated Banks (1990) study found significant differences in wing and bill size between eastern and western cuckoos, but it still concluded that the subspecies should not be recognized. Franzreb and Laymon (1993) used Banks's data and determined that there were significant differences between eastern and western cuckoos in wing, tail, and bill lengths, as well as bill depth, in addition to potential behavioral, vocal, and ecological differences. Franzreb and Laymon (1993) concluded that recognition of the two subspecies should be retained until further examination determined otherwise. The two subspecies are separated by geographic distribution, with the boundary between the two subspecies considered to be the Pecos River in Texas (Hughes 1999). It should be noted that the USFWS refers to the western U.S. DPS in the October 2011 annual review (76 FR 66370-66439) rather than the state-listed western yellow-billed cuckoo subspecies.

Descriptions of the species' physical characteristics can be found in Hughes (1999).

Distribution

General

The western yellow-billed cuckoo's historical geographic range is southwestern British Columbia, western Washington, northern Utah, central Colorado, western Texas, south and west to California, and southern Baja California, Sinaloa, and Chihuahua in Mexico (Hughes 1999) (Figure SP-B15). The western yellow-billed cuckoo is rare and local in the southwestern United States. It breeds along the major river valleys in southern and western New Mexico, and central and southern Arizona. In California, the western yellow-billed cuckoo's breeding distribution is now thought to be restricted to isolated sites in the Sacramento, Amargosa, Kern, Santa Ana, and Colorado River valleys (Laymon and Halterman 1987). During surveys in 1999 and 2000 western yellow-billed cuckoos were not found on the Amargosa and Santa Ana rivers (Laymon, pers. comm. 2012).

Distribution and Occurrences within the Plan Area

Historical

The California Natural Diversity Database (CNDDB) contains 28 historical (i.e., pre-1990) occurrence records dating from 1917 to 1986. Of the known occurrences, 24 are from 2 years: 1977 (13)and 1986 (11). Single known occurrences are from 1917, 1945, 1978, and 1983. Of the historical known occurrences in the Plan Area, 23 are from the Lower Colorado River, with 14 known occurrences from Imperial County, ranging the Palo Verde area to the U.S.–Mexico border; 6 from eastern Riverside County in the Blythe area; and 2 from San Bernardino County in the Needles area. Five of the historical known occurrences are from the Amargosa River, Tecopa, China Ranch, and Independence areas in Inyo County, and 2 are from the Mojave River in the Upper Narrows and Hodge areas in San Bernardino County. Of 28 historical known occurrences, the majority are on public land.

Recent

In the Sacramento Valley, the south coast (including Ventura and Los Angeles counties), and Kern County, yellow-billed cuckoos were considered common to numerous in late the 1800s, but only fairly common by 1920s (Hughes 1999). By the 1950s, the subspecies had been extirpated north of Sacramento Valley (Hughes 1999). The species may also no longer breed in the Amargosa and Santa Ana rivers (Laymon, pers. comm. 2012).

The CNDDB contains nine recent (i.e., since 1990) occurrences for the Plan Area: a 1991 known occurrence in the Alabama Hills near Lone Pine, a 1998 known occurrence from the Laguna Dam area of the Colorado River in Imperial County, a 2009 occurrence north of the Cibola National Wildlife Refuge (NWR), a 2009 occurrence in the Imperial NWR area, and three 2009 occurrences along the Colorado River in the Palo Verde Ecological Reserve in Riverside County (Figure SP-B15) (CDFW 2013).

Natural History

Habitat Requirements

This discussion is limited to breeding habitat requirements for western yellow-billed cuckoo in California. Breeding habitat primarily consists of large blocks, or contiguous areas, of riparian habitat, particularly cottonwood–willow riparian woodlands (66 FR 38611–38626) (see Table 1). From a survey conducted from northern Kern and Inyo counties south in 1986 and from southern Kern and Mono counties north in 1987, Laymon and Halterman (1989) proposed that optimum habitat patches for the western yellow-billed cuckoo are greater than 200 acres in size and wider than 1,950 feet; sites 101 to 200 acres in size and wider than 650 feet were suitable; sites 50 to 100 acres in size and 325 to 650 feet were marginal; and sites smaller than these dimensions were unsuitable. Western yellow-billed cuckoo prefers dense riparian thickets with dense low-level foliage near slow-moving water sources. Nests are constructed in willows on horizontal branches in trees, shrubs, and vines, but cottonwoods (*Populus* spp.) are used extensively for foraging, and humid lowland forests are used during migration (Hughes 1999). Of 95 detected nests at the South Fork Kern River, all were in willows, with one exception in a cottonwood (Laymon 1998). Along the Santa Ana River, 92% of nests were in willows, with one nest in a mistletoe clump in a cottonwood and one in an alder (*Alnus* spp.) (Laymon 1998). Nests along the Sacramento River have been found in willow, cottonwood, and alder, and also, although rarely, in orchards (Laymon 1998).

Laymon (1998) presents some detailed habitat information for the Bill Williams River in the Lake Havasu area in Arizona. This area is the most relevant to the Plan Area populations in the lower Colorado River area. Of 14 nests detected in the Bill Williams River, 11 were in willows, 1 in a cottonwood, and 2 in tamarisk (*Tamarix* spp.). Canopy closure averaged 77% and range from 51% to 92%; shrub averaged 33% with a range of 5% to 85%. The average distance of nests to water was 135 feet with a range of 0 to 575 feet.

Table 1. Habitat Associations for Western Yellow-Billed Cuckoo

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Riparian woodland and forest	Nesting and foraging	Primary	Patch size > 198 acres; width > 1,270 feet; dense vegetation	Laymon and Halterman 1989

Foraging Requirements

Yellow-billed cuckoos generally forage for lepidopteran larvae (caterpillars) and other large insects such as katydids by gleaning (Hughes 1999; Laymon 1998). They will also occasionally prey on small lizards, frogs, eggs, and young birds (Gaines 1999; Laymon 1998). Foraging occurs extensively in cottonwood riparian habitat (Hughes 1999).

Reproduction

In the western United States, nests are typically constructed in willows (*Salix* spp.), Fremont cottonwood (*Populus fremontii*), mesquite (*Prosopis* spp.), hackberry (*Celtis* spp.), soapberry (*Sapindus saponaria*), alder (*Alnus* spp.), or cultivated fruit trees on horizontal branches or vertical forks of the large tree or shrub (Hughes 1999). Nests are generally placed between 1 and 6 meters (3 and 20 feet) above the ground and concealed by foliage, especially from above (Hughes 1999). Nest sites in arid regions are restricted to relatively humid river bottoms, ponds, swampy areas, and damp thickets (Hughes 1999). Both the male and female build the nest from twigs (approximately 15 centimeters [6 inches] long) likely collected within 10 meters (33 feet) of the nest site (Hughes 1999).

The western yellow-billed cuckoo has a short breeding season, lasting only about 4 months from time of arrival on breeding grounds in the spring to fall migration (see Table 2). Western yellow-billed cuckoos typically lay a single clutch per season of two or three eggs (average is just over two eggs, and up to four eggs per clutch is known) in mid-June to mid-July, and incubation occurs over 9 to 11 days (Hughes 1999; Johnson et al. 2008). However, Laymon (1998) reports

that in years of abundant resources, double- and even triple-clutching in a season can occur along the South Fork Kern River; over a 12-year period, double-clutching occurred less than half of the study years, and triple-clutching only occurred one year. Double-clutching has not been observed at the Bill Williams River site near the Colorado River (Laymon 1998). Development of the young is very rapid, with fledging occurring in 6 to 9 days; the entire breeding cycle may be only 17 days from egg laying to fledging of the young (Hughes 1999). Fledglings are dependent upon parents for up to 3 weeks following fledging (Johnson et al. 2008). Females often switch mates between broods within years and usually select a new mate in subsequent years. They can also be communal nesters with 2 females laying eggs in a nest and tending the young. Nests often have a helper male that tends the young (Laymon, pers. comm. 2012). The yellow-billed cuckoo has been noted to be both an intraspecific and interspecific brood parasite (Hughes 1999); however, this appears to only occur in the eastern yellow-billed cuckoo. The western yellow-billed cuckoo apparently is rarely parasitized by the brown-headed cowbird (*Molothrus ater*), possibly because its short breeding period reduces the chance of successful nest parasitism (Hughes 1999).

Table 2. Key Seasonal Periods for Western Yellow-Billed Cuckoo

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Breeding					X	X	X	X	X			
Migration									X	X		

Notes: Breeding in late May is rare.
Sources: Laymon 1998; Hughes 1999; Gaines 1999.

Spatial Behavior

Spatial behavior patterns in the western yellow-billed cuckoo include migration, territory use, and dispersal from natal sites, as summarized in Table 3.

The western yellow-billed cuckoo is a long-distance migrant, although details of its migration patterns are not well known (Hughes 1999). It

is a relatively late spring migrant, arriving on the breeding grounds starting mid- to late May, but more commonly in June, and leaving from late August to early September (Franzreb and Laymon 1993; Gaines 1999) (Table 2). The migratory route of the western yellow-billed cuckoo is not well known because few specimens collected on wintering grounds have been ascribed to the western or eastern subspecies. The western yellow-billed cuckoo likely moves down the Pacific Slope of Mexico and Central America to northwestern South America (Hughes 1999).

Western yellow-billed cuckoos may have variable breeding territory sizes, with territories reported to be as small as 10 acres on the Colorado River (Laymon and Halterman 1989), but with a range of 20 to 100 acres on the South Fork Kern River (Laymon 1998). Recent data from radio telemetry studies on the Colorado, San Pedro, and Rio Grande rivers have shown larger home ranges. Cuckoos on the Rio Grande in New Mexico used an average of 204 acres (Sechrist et al. 2009), while cuckoos on the San Pedro River in Arizona, averaged about 125 acres (Halterman 2009). On the Colorado River in Arizona and California, cuckoos home ranges averaged about 95 acres (McNeil et al. 2010; McNeil et al. 2011a, 2011b). Whether western yellow-billed cuckoos are "territorial" in the sense of defending a spatially defined area is uncertain, although individuals have been observed to aggressively supplant each other (Hughes 1999).

Dispersal and the degree to which the western yellow-billed cuckoo shows site fidelity is largely unknown. The absence of pairs on known breeding sites in some years and presence of breeding birds on previously vacant sites demonstrates that breeding may not occur in the same location every year (Gaines and Laymon 1984). However, some breeding pairs along the South Fork Kern River have returned to the same nest territories for up to 3 years (unpublished data reported by Laymon 1998). Limited banding data indicate birds returning to breeding sites within 1.2 miles of natal sites (Hughes 1999), but too few birds have been banded and monitored to document typical dispersal patterns with any confidence. Along the South Fork Kern River, all banded individuals that have been resighted in the same area have been males (Laymon 1998).

Table 3. Spatial Behavior by Western Yellow-Billed Cuckoo

Type	Distance/Area	Location of Study	Supporting Information
Home Range (Territory?)	As small as 10 acres	Colorado River	Laymon and Halterman 1989
Home Range	20–100 acres	South Kern River	Laymon 1998

Ecological Relationships

Intraspecific and interspecific and community relationships are not well understood for the western yellow-billed cuckoo. The eastern yellow-billed cuckoo is an intraspecific and interspecific brood parasite, but this behavior has not been documented in the western yellow-billed cuckoo (Hughes 1999). Where brood parasitism does occur, yellow-billed cuckoos may be mobbed and harassed by other native birds such as American robin (*Turdus migratorius*) (Hughes 1999). Otherwise, there is no information regarding intraspecific and interspecific relationships or competition (Hughes 1999).

Western yellow-billed cuckoos are vulnerable to predation by other birds, particularly by raptors during migration, snakes, and mammals (Hughes 1999). Laymon (1998) reports that red-shouldered hawk (*Buteo lineatus*) and northern harrier (*Circus cyaneus*) have preyed on nestlings and that cuckoos chase western scrub-jay (*Aphelocoma californica*) and loggerhead shrike (*Lanius ludovicianus*) away from nests.

Presence and successful breeding by yellow-billed cuckoos may be limited by available resources. At occupied breeding sites, nesting success may be limited by available food sources. Cuckoo chicks hatch asynchronously, so the nest may contain unhatched eggs and young of various ages (Hughes 1999). The youngest chick in a brood may not be fed when food sources are in short supply, and birds may not reproduce at all when insufficient food is available (Hughes 1999). It also appears that increased food availability has a positive effect on clutch size (Martin 1987; Laymon 1998). A study of the effects of climate on yellow-billed cuckoo found that nesting by eastern yellow-billed cuckoos in the 2003 and 2004 breeding seasons only occurred at

sites where caterpillars were more abundant (Anders and Post 2006) (also see discussion below on climate effects). Laymon (1998) reports that western yellow-billed cuckoos may produce multiple clutches along the South Fork Kern River when food sources are abundant.

Population Status and Trends

Global: Declining (NatureServe 2010)
State: Declining (Laymon 1998)
Within Plan Area: Same as above

Western yellow-billed cuckoo was once considered common to numerous in the Sacramento Valley, along the southern coast of California from Ventura to Los Angeles counties, and in Kern County in the late 1800s, but it was considered only fairly common by the 1920s (Gaines 1974; Gaines and Laymon 1984). The numbers of yellow-billed cuckoos in California and other western areas had declined markedly into the 1980s with loss of riparian habitats (Laymon and Halterman 1987). Surveys in 1986 and 1987 showed a decline from 123 to 163 pairs in 1977 to 30 to 33 pairs in 1987, or a 73% to 82% decline over this 10-year period (Laymon 1998). The most recent statewide surveys in 1999 and 2000, including the Sacramento, Kern, and Lower Colorado rivers (1999 only), as well as other areas with smaller amounts of habitat, documented 41 to 45 pairs and 49 unmated birds in 1999, and 61 to 67 pairs and 61 to 68 unmated birds in 2000 on the Sacramento and Kern rivers (Halterman et al. 2003). Although the number of detected pairs was higher in 1999-2000 compared to 1986-1987, there were still substantially fewer pairs than detected in 1977.

The western yellow-billed cuckoo suffered substantial range reductions in the twentieth century due to loss of riparian habitat (Laymon and Halterman 1987). The species was extirpated north of Sacramento Valley by the 1950s (Gaines and Laymon 1984). Surveys throughout California in 1986–1987 found that only three areas in the state supported more than approximately five breeding pairs on a regular basis, including the Sacramento River between Colusa and Red Bluff, the South Fork of the Kern River, and the lower Colorado River (Johnson et al. 2008). In the 1999-2000 surveys, the Sacramento and Kern rivers were the only remaining areas with

more than 1,000 hectares (2,470 acres) each of prime suitable habitat (i.e., high canopy cover, extensive understory, and structural diversity) (Halterman et al. 2003).

Within the Plan Area, the majority of CNDDB records are from the Colorado River (CDFW 2013). Once considered abundant throughout the lower Colorado River, a dramatic decline of the species was noted during surveys in the 1970s and 1980s. The lower Colorado River and its tributaries supported an estimated 180–240 pairs in 1976–77. This population declined by an estimated 80% to 90% by 1986. In 1998, no pairs could be identified west of the Colorado River in the parts of California that had been occupied in 1976–77. Along the lower Colorado River and its major tributaries, losses have been greatest at lower elevations below 900 meters (3,000 feet) (Johnson et al. 2008).

Threats and Environmental Stressors

The western yellow-billed cuckoo is sensitive to habitat fragmentation and degradation of riparian woodlands due to agricultural and residential development (Hughes 1999), and major declines among western populations reflect local extinctions and low colonization rates (Laymon and Halterman 1989). Groundwater pumping and the replacement of native riparian habitats by invasive non-native plants, especially tamarisk, have substantially reduced the area and quality of available breeding habitats for yellow-billed cuckoo (75 FR 69222–69294). Even where habitat is not degraded, the species has been extirpated from breeding areas occupied by four or fewer pairs (Laymon and Halterman 1987), possibly due to the inherent instability of small populations (Laymon and Halterman 1989). The extensive surveys in 1999 and 2000 found that large breeding populations in California only remain on the Sacramento and Kern rivers where there is still substantial prime habitat (Halterman et al. 2003). Non-native invasive species such as tamarisk (*Tamarix* spp.) may preclude use by western yellow-billed cuckoos; previously occupied willow–cottonwood habitats that converted to monotypic stands of tamarisk generally were no longer inhabited (Laymon and Halterman 1987), although Laymon (1998) reports two nest sites in tamarisk at the Bill Williams River site in Arizona. However, even at these sites, the habitat within the cuckoos' territories was still primarily willow-cottonwood (Laymon, pers. comm. 2012). Of the 33

known occurrences in the CNDDB database for the Plan Area, three of the sites were reported to have tamarisk invasion (CDFW 2013).

Pesticides may affect behavior of western yellow-billed cuckoo by loss of balance or may cause death by direct contact (Hughes 1999). Pesticides may contaminate preferred prey items, particularly lepidopteran larvae. In addition, some prey species, such as frogs, occur in pesticide-laden runoff adjoining agricultural land (Laymon and Halterman 1987). The western yellow-billed cuckoo also has shown pesticide effects on reproduction due to eggshell thinning (Gaines and Laymon 1984; Laymon and Halterman 1987). Of the 33 known occurrences in the Plan Area, agriculture (and associated access roads) adjacent to occupied habitat was reported to be a threat to five of the sites (CDFW 2013).

Yellow-billed cuckoos are also known to collide with windows, resulting in injuries and fatalities (Klem 1989, 1990). Whether this a substantial threat in the Plan Area is unknown, but it seems unlikely given the limited amount of development in occupied areas.

Climate change may be a stressor on yellow-billed cuckoos. Anders and Post (2006) examined BBS data for the eastern yellow-billed cuckoo for the period of 1966 to 2002 in relation to the North American Oscillation and El Niño Southern Oscillation climate systems. (The western yellow-billed cuckoo was excluded from the analysis due to few data.) Anders and Post (2006) found that populations were sensitive to warm temperatures, with population declines in the year following the preceding breeding season with warm temperatures. They postulate that the decline in productivity is related to reduced available prey because they found that breeding only occurred in 2003 and 2004 on sites with more abundant prey. Lepidopteran larvae outbreaks appear to be more common during cooler weather (Anders and Post 2006). Further, it is possible that warmer temperatures cause earlier peaks of lepidopteran larvae that could be asynchronous with breeding by yellow-billed cuckoos at a time when prey is needed most (Anders and Post 2006).

Conservation and Management Activities

A rangewide conservation and assessment strategy for the western yellow-billed cuckoo is currently in preparation by a group of federal, state, and nongovernmental agencies organized by the Sacramento office of the USFWS (75 FR 69222–29294). Work on the conservation strategy is expected to be initiated in 2011.

Known occurrences of western yellow-billed cuckoo in the Plan Area are on BLM land. BLM Manual 6840 establishes Special-Status Species policy for plant and animal species and the habitat on which they depend (BLM 2001). The objectives of the BLM policy are:

A. To conserve listed species and the ecosystems on which they depend.

B. To ensure that actions requiring authorization or approval by the BLM are consistent with the conservation needs of special-status species and do not contribute to the need to list any special-status species, either under provisions of the ESA or other provisions of this policy (BLM 2001).

The BLM has identified the western yellow-billed cuckoo as a sensitive species and requires surveys in suitable habitat areas prior to authorizing activities that could disturb the species or its habitat.

Although the western yellow-billed cuckoo is not federally listed, several habitat conservation plans that would provide regulatory coverage for species, were it to be listed, have been implemented, including the Clark County Nevada Habitat Conservation Plan; the Lower Colorado River Multi-Species Conservation Plan; and the California Department of Corrections Electrified Fence Project (for 26 sites throughout California, including nine sites in the Plan Area). Each of these conservation plans provides for conservation/protection and management of habitats that benefit the western yellow-billed cuckoo.

Wetland permits under Section 1600 of the California Department of Fish and Wildlife Code and federal Clean Water Act 404 issued by CDFW and the U.S. Army Corps of Engineers, respectively, also typically require avoidance, minimization, and mitigation measures for impacts to riparian habitats that may be used by western

yellow-billed cuckoo and which may benefit the species. Further, any impacts to the species resulting in "take" are regulated by Section 2081 of the California Endangered Species Act, and full mitigation of impacts is required.

Data Characterization

Statewide systematic surveys for the western yellow-billed cuckoo have not been conducted since 1999 and 2000 (Halterman et al. 2003), and there are only three recent (since 1990) known occurrences in the CNDDB for the Plan Area (CDFW 2013). The current status of the species along the lower Colorado River and other areas where it has historically occurred, such as the Amargosa and Mojave rivers, is unknown. However, 26 of the 33 historic and recent known occurrences of the species are on public lands and are not subject to intense development pressure. The main concern for these areas is current habitat quality given that the western yellow-billed cuckoo requires large, dense tracts of riparian habitat. Water development (e.g., in the Victorville area) and invasive species such as tamarisk may have caused habitat degradation at some of the known occurrence sites since the cuckoo has been seen in the areas.

Management and Monitoring Considerations

Western yellow-billed cuckoo usually occur in large, dense tracts of riparian habitat, as summarized previously under Habitat Requirements. Therefore, management and monitoring will need to focus on maintaining, restoring, and enhancing large tracts of suitable habitat for the species, including controlling invasive species, such as tamarisk (Laymon and Halterman 1985; Laymon 1998; Sogge et al. 2008) and ensuring water sources to maintain large riparian areas. The native, deep-rooted species that compose suitable cuckoo habitat, generally associated with perennial watercourses, require floods for maintenance and are tolerant of submersion when young (66 FR 38611–38626; Hughes 1999). Fire is also a consideration along the Colorado River, especially where people camp and may leave unattended camp fires (Comrack, pers. comm. 2011). The species is also highly dependent on adequate food sources (primarily caterpillars) for successful breeding (Martin 1987; Hughes 1999;

Anders and Post 2006), so potential impacts on the prey base by pesticides applied to agricultural areas near suitable habitat are also a management concern. Pesticides may also cause lethal and sublethal poisoning to adults and young, adversely affecting the health and reproductive fitness of individuals and the viability of populations (Hughes 1999).

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for western yellow-billed cuckoo, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 174,654 acres of modeled suitable habitat for western yellow-billed cuckoo in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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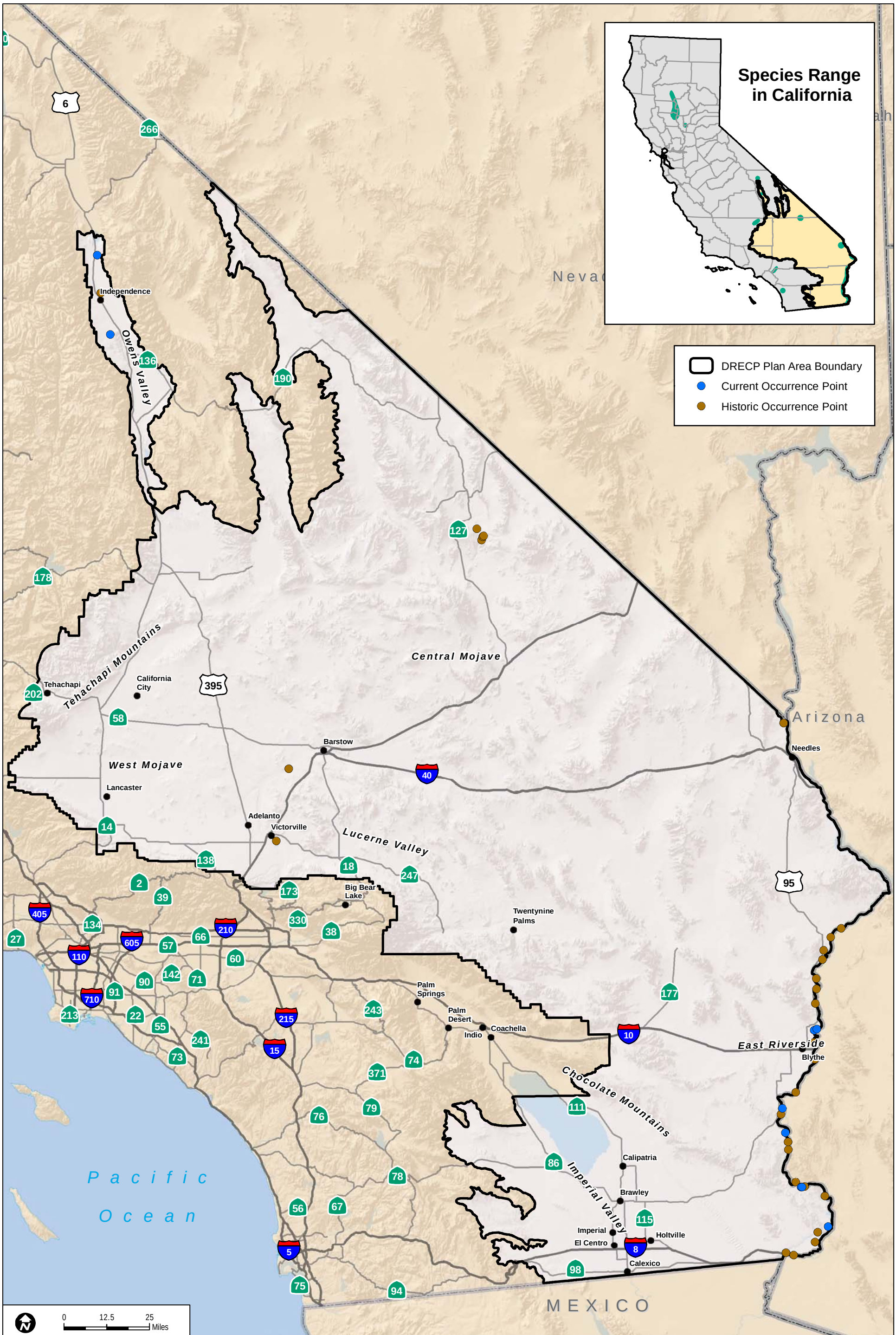

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B13

Western Yellow-billed Cuckoo Occurrences in the Plan Area

Willow Flycatcher (*Empidonax traillii*)

Photo by Dudek.

Legal Status

State: Endangered (willow flycatcher full species)
Federal: Endangered (southwestern willow flycatcher subspecies)
Critical Habitat: Designated on October 19, 2005 (70 FR 60886–61009) for southwestern willow flycatcher. The U.S. Fish and Wildlife Service (USFWS) proposed revised critical habitat on August 15, 2011 (76 FR 50542-50629), but the 2005 designation is still in place pending issuance of a final rule.
Recovery Planning: Final recovery plan (USFWS 2002) for southwestern willow flycatcher

Taxonomy

The willow flycatcher (*Empidonax traillii*) is a small passerine that was once considered along with the alder flycatcher (*E. alnorum*), as Traill's flycatcher (Grinnell and Miller 1944). Since 1973 the American Ornithological Union (AOU) has treated the alder flycatcher as a separate species and there are currently four recognized subspecies of *E. traillii*, three of which occur in California (*E. t. brewsteri*, *E. t. adastus*, and *E. t. extimus*) (USFWS 2002; Unitt 1987). Only the southwestern willow flycatcher subspecies (*E. t. extimus*) breeds in the Plan Area, and it is the primary focus of this account. The other two subspecies occur in the Plan Area only briefly during migration, and they are addressed in this account where relevant. The southwestern willow flycatcher was described by A. R. Phillips in 1948 from a collection by G. Monson from the lower San Pedro River in southwestern Arizona (60 FR 10695–10715). Southwestern willow flycatcher can be phenotypically distinguished from the other subspecies by its paler color, wing ratio, and song dialect (60 FR 10695–10715), although these are not reliable field identification

criteria (Sogge, pers. comm. 2012). Paxton (2000) concluded that the *E. t. extimus* subspecies is genetically distinct from the other subspecies, although intergrades between *E. t. adastus* and *E. t. extimus* have been reported (Unitt 1987).

Distribution

General

The willow flycatcher occurs throughout the United States with the exception of the extreme northeast and the southeast. In California, breeding populations of *E. t. adastus* and *E. t. brewsteri* are separated by the crest of the Sierra Nevada, while the historical range of *E. t. extimus* includes riparian habitats in the southern one-third of California, southern Nevada, Arizona, New Mexico, western Texas and northern Mexico (Sogge et al. 2010; USFWS 2002; Figure SP-B13), and, again, this is the only subspecies breeding in the Plan Area. The current range of *E. t. extimus* is similar to its historical range, the main difference being a reduction in the distribution and amount of existing suitable habitat within its historical range. This subspecies' breeding range extends as far north as the Santa Ynez River, Kern River, and the town of Independence on the Owens River (Craig and Williams 1998). Outside of California, historical breeding has occurred in southern Nevada, southern Utah, Arizona, New Mexico, and southwestern Colorado (Paxton 2000; Sogge et al. 2010).

Distribution and Occurrences within the Plan Area

Historical

Within the Plan Area, breeding southwestern willow flycatchers have been found at five general locations: Owens River Valley, Mojave River, San Felipe Creek (a tributary of the Salton Sea), the Lower Colorado River between Hoover and Parker, and the Lower Colorado River between Parker and the international boundary (Durst et al. 2008a). Willow flycatcher populations at these locations still exist, although numbers of territories have greatly declined at some locations, especially along the Colorado River (Durst et al. 2008a). These sites are discussed in further detail in the following section.

There are no known general locations in the Plan Area that previously supported, but no longer support, southwestern willow flycatchers.

There are four historical (i.e., pre-1990) occurrences for southwestern willow flycatcher recorded in the Plan Area (CDFW 2013; Dudek 2013). The southwestern willow flycatcher occurrences are located north of Independence in Inyo County and in the vicinity of Mojave and California cities (Figure SP-B13).

Recent

As mentioned previously, there are five general locations in the Plan Area that currently support breeding populations of southwestern willow flycatchers. However, the southwestern willow flycatcher exhibits metapopulation dynamics with individuals commonly moving both among different sites within a breeding area and among different breeding areas (Sogge et al. 2010). Such movements reflect the dynamic interaction of suitable habitat and selection of breeding sites. In particular, small breeding sites are subject to variable use (Sogge, pers. comm, 2012). A detailed discussion of each of the five general breeding locations follows.

Owens River Valley: Most recently (as of 2007), Durst et al. (2008a) identified 28 territories at five sites in the Owens River Valley. However, almost all these territories occur north of the Plan Area. Within the Plan Area, two territories were located along the Owens River near Lone Pine in 1999, but the current breeding status at this location is unknown. Rourke et al. (2004) surveyed Hogback Creek near Lone Pine in 2001, but found no southwestern willow flycatchers. It is possible that none of the extant southwestern willow flycatcher territories found in the Owens River Valley occur within the Plan Area.

Mojave River: Durst et al. (2008a) stated that as of 2007, four nesting territories occur along the Mojave River near Victorville, but that territories are now gone from at least three other sites (Oro Grande, Upper Narrows, and Victorville Interstate 15). Nearby Holcomb Creek also once supported nest territories.

San Felipe Creek: San Felipe Creek is a tributary of the Salton Sea and as of 2007 supported four southwestern willow flycatcher nesting territories (Durst et al. 2008a).

Lower Colorado River – Hoover to Parker: As of 2007, Durst et al. (2008a) identified 14 territories remaining at six sites along this stretch of the Colorado River. However, most of these territories occur at Topock Marsh on the Arizona side of the border. A California territory at Trampas Wash is considered extirpated (Durst et al. 2008a).

Lower Colorado River – Parker to South International Border: At one time, breeding southwestern willow flycatchers were located at 16 sites along this stretch of the Lower Colorado River, mostly on the Cibola and Imperial National Wildlife Refuges (NRWs). By 2007, the number of territories was reduced to one. McLeod and Koronkiewicz (2009) resurveyed this stretch in 2008 and "rediscovered" some territories (e.g., at Big Hole Slough), but territory numbers remain very low.

There are 101 recent (i.e., since 1990) occurrence records for willow flycatcher, of which the vast majority are identified only as willow flycatcher (CDFW 2013; Dudek 2013). There are five recent records for southwestern willow flycatcher along the Lower Colorado River in the stretch between the Cibola and Imperial NWRs, just south of where Interstate 10 crosses the river, and in the Havasu NWR area. There are also recent occurrences for southwestern willow flycatcher north of Niland east of the Salton Sea, in the Mojave River Narrows Regional Park, and in a tributary to the Owens River just above Tinemaha Reservoir. The remaining recent willow flycatcher occurrences are located in several regions of the Plan Area, including: Ridgecrest and the China Lake Naval Air Weapons Station, Amargosa Canyon, the Fremont Valley in the western Mojave, the southern Sierra Foothills west of Red Rock Canyon State Park, the Cities of Mojave and California City, Galileo Park north of 20 Mule Team Parkway, the southwestern portion of Edwards Air Force Base, the western portion of Mojave National Preserve, the Kingston Range, the Morongo Valley, Lake Tamarisk Golf Course in the Chuckwalla Valley, and north of Niland east of the Salton Sea.

Natural History

Habitat Requirements

In California, the southwestern willow flycatcher is restricted to riparian habitats occurring along streams or in meadows (Craig and Williams 1998; Sogge et al. 2010). As noted above under Distribution and Occurrences, there is a dynamic relationship between suitable habitat and selection of breeding sites, with individuals commonly moving within general breeding areas and among different breeding areas. The structure of suitable breeding habitat typically consists of a dense mid-story and understory and can also include a dense canopy (60 FR 10695–10715). However, suitable vegetation is not uniformly dense and typically includes interspersed patches of open habitat. Typical plant species associated with their habitat include willow (*Salix* spp.), mulefat (*Baccharis salicifolia*), stinging nettle (*Urtica* spp.), cottonwood (*Populus* spp.), tamarisk (*Tamarix* spp.), and Russian olive (*Elaeagnus angustifolia*). Within the habitat structure parameters discussed above, southwestern willow flycatcher does demonstrate adaptability in that it can occupy riparian habitats composed of native broadleaf species, a mix of native and exotic species, or monotypic stands of exotics (Sogge et al. 2010). This subspecies is known to nest in monotypic stands of Russian olive and tamarisk (60 FR 10695–10715). Furthermore, along the San Luis Rey River in San Diego County, southwestern willow flycatcher has nested in riparian habitat dominated by coast live oak (*Quercus agrifolia*), and in Cliff‑Gila Valley in New Mexico they are known to nest in tall box-elder. Plant species composition does not seem as important as a dense twig structure and an abundance of live, green foliage (Sogge et al. 2010). Also, the location of the nest seems to depend more on suitable twig structure and live vegetative cover than height or plant species composition (Sogge et al. 2010).

Riparian habitats within the Plan Area are also important stopovers to *E. t. adastus* and *E. t. brewsteri* as they migrate through (Finch and Kelley 1999). However, during migration willow flycatchers also use non-riparian habitats, including shrublands, grasslands, and agriculture (Finch et al. 2000). Other habitats used during migration typically lack

the features associated with breeding sites, such as standing water, moist soils, and patch size and structure (Finch et al. 2000).

Southwestern willow flycatcher nesting sites are generally located near surface water or saturated soils (Table 1). Due to the variability of hydrologic conditions in Southern California, water availability at a site may range from inundated to dry from year to year or within the breeding season. Nonetheless, moisture levels must remain high enough to support appropriate riparian vegetation (Sogge et al. 2010). Dense willow thickets are the most important habitat component for breeding *E. t. adastus* and *E. t. brewsteri* in California (Stefani et al. 2001).

Table 1. Habitat Associations for Southwestern Willow Flycatcher

Land Cover Type	Land Cover Use	Habitat Designation	Habitat Parameters	Supporting Information
Dense Riparian	Breeding	Primary	Dense understory and mid-story	60 FR 10695–10715
Riparian	Foraging	Secondary	Openings within and edges of breeding habitat, over wet areas	Finch and Stoleson 2000

Foraging Requirements

Southwestern willow flycatchers are insectivorous and forage at the edges or internal openings of their territory, above the canopy or over open water. There are records of adults foraging outside of their territory and even within neighboring territories (Finch and Stoleson 2000). Their diet consists mainly of bees, wasps, flies, leaf hoppers, and beetles (Durst et al. 2008b), which they catch in the air, glean from vegetation, or occasionally pick, catch, or seize from the ground (Sedgwick 2000). However, because southwestern willow flycatcher is a generalist, its specific diet is difficult to describe. Diets can vary depending on the breeding site and weather conditions (Durst et al. 2008b). Presumably, the diet of migrating *E. t. adastus* and *E. t. brewsteri* is similar.

Reproduction

Southwestern willow flycatcher males and females become reproductively viable during their second year. This subspecies is predominantly monogamous although reports of polygyny are not uncommon (Sedgwick 2000). Males arrive at the breeding sites between early May and early June (USFWS 2002; Table 2). Females arrive 1 to 2 weeks after males and inhabit the territory of a male (Finch and Stoleson 2000). Nest building begins approximately 2 weeks after pair formation. Females build an open cup nest measuring 8 centimeters high by 8 centimeters wide (3.1 by 3.1 inches) with little to no assistance from the male.

The female incubates the eggs for an average of 12 to 13 days. The female provides the majority of care for the young; however, the male becomes more involved as the nestlings grow and demand more food. The nestlings fledge between 12 and 15 days after hatching (Sogge et al. 2010).

Southwestern willow flycatcher will typically renest following an unsuccessful attempt and less frequently may renest following a successful attempt. The clutch size of the first nesting attempt is typically three to four eggs but decreases with each new attempt (Ellis et al. 2008).

Studies in California along the South Fork Kern River showed that site fidelity for banded adults was 35.8% (Craig and Williams 1998); however, these studies did not differentiate between site fidelity and mortality. Studies in Arizona that only included surviving adults showed site fidelity as high as 66% as opposed to less than 50% for studies in the same area that did not take mortality into consideration (Luff et al. 2000). As *E. t. adastus* and *E. t. brewsteri* do not breed in the Plan Area, they are not addressed in this section.

Table 2. Key Seasonal Periods for Southwestern Willow Flycatcher

	Jan	Feb	March	April	May	June	July	Aug	Sep	Oct	Nov	Dec
Arrival					X	X						
Breeding					X	X						
Fledges						X	X					
Migration South								X	X			

Sources: 60 FR 10695–10715; USFWS 2002

Spatial Behavior

During their northbound and southbound migrations, other subspecies of willow flycatcher pass through areas occupied by nesting southwestern willow flycatchers. In Southern California, peak numbers of northbound *E. t. brewsteri* migrate the first couple weeks of June through occupied *E. t. extimus* breeding territories (Finch and Stoleson 2000). Therefore, for the purpose of focused surveys for southwestern willow flycatcher, willow flycatchers occurring within the southwestern willow flycatcher breeding range can only be assumed to be southwestern willow flycatcher if detected between June 15 and July 20, when *E. t. brewsteri* have passed north to their breeding grounds (USFWS 2002). Willow flycatchers in the southwest migrate along riparian corridors (Finch and Stoleson 2000); because all three subspecies in California seasonally occur both north and south of the Plan Area, any riparian habitat within the Plan Area might represent important migration habitat for willow flycatchers. Finch and Kelley (1999) found that while migrating along the Rio Grande, willow flycatchers (including *E. t. extimus*) preferred habitats dominated by willows over other riparian species.

In adult southwestern willow flycatchers, movement to different breeding sites from year to year is not an uncommon occurrence and may occur as a response to low reproductive success at a particular nesting site. Distances covered range from 0.1 to 214 kilometers (0.06 to 133 miles) (Table 3). Year to year dispersal among juvenile birds is higher than in adults because juveniles rarely return to their natal site (Paxton 2007). Movement between breeding sites within the same

breeding season typically occurs during pre- or post-breeding; although territory switching does occur, it makes up a small percentage of this type of movement (Paxton et al. 2007).

Territory sizes vary greatly depending on several factors, including but not limited to quality of habitat and population density. The observed range of territory sizes is about 0.1 to 2.3 hectares (0.3 to 5.7 acres), with most in the range of 0.2 to 0.5 hectares (0.5 to 1.2 acres) (USFWS 2002). Male territories tend to be larger before and after breeding. The area utilized within a territory tends to be smallest during incubation and when occupied by nestlings (Sogge et al. 2010).

Wintering locations for southwestern willow flycatcher are becoming better understood. Paxton et al. (2011a) combined information from mitochondrial DNA sequences and morphological characteristics from museum specimens collected for willow flycatchers from across their winter range and found that the Pacific lowlands of Costa Rica appear to be a key winter location for southwestern willow flycatcher, although Central American countries may also be important for the subspecies. Willow flycatchers will travel between 3,200 and 8,000 kilometers (2,000 and 5,000 miles) round-trip from their wintering sites to their breeding sites. During migration, willow flycatchers use a greater variety of habitats, including some with non-riparian vegetation (Finch and Stoleson 2000).

Table 3. Movement Distances for Southwestern Willow Flycatcher

Type	Distance/Area	Location of Study	Citation
Breeding Territory	0.1–<2.3 hectares	California	USFWS 2002
Dispersal	0.1–214 kilometers	Arizona	Paxton 2007
Migration	3,200–8,000 kilometers	Throughout range	Finch and Stoleson 2000

Ecological Relationships

As is common for passerine bird species, southwestern willow flycatcher juveniles, eggs, and (less often) adults, are preyed upon by other birds, mammals, and reptiles. Predation is often the main factor

responsible for nest failure (Sogge et al. 2010). In studies conducted along the lower Colorado River in 2003, depredation accounted for 57% of all documented nest failures (Koronkiewicz et al. 2004).

Brown-headed cowbirds (*Molothrus ater*), which are obligate brood parasites, parasitize the nests of several native passerine species, including southwestern willow flycatcher, and therefore also contribute to the overall nest failure for this subspecies. Female cowbirds lay their eggs in the nests of other bird species (host pair) at the expense of the reproductive success of the host pair (Finch and Stoleson 2000). Cowbirds have existed sympatrically with southwestern willow flycatcher throughout most of its range for hundreds of thousands of years. However, in Southern California, these two species have only co-occurred since 1900 (USFWS 2002). Nonetheless, the defense mechanisms used by southwestern willow flycatcher in Southern California in response to nest parasitism are similar to those used by willow flycatchers elsewhere, including nest abandonment (USFWS 2002) or burying the parasite egg in the nest floor (Finch and Stoleson 2000). Most southwestern willow flycatchers renest after abandoning their nest due to parasitism (USFWS 2002) and do not typically fledge flycatcher young from a parasitized nest (Sogge et al. 2010).

Despite evidence for parasitism, brown-headed cowbirds are not considered a primary threat to the success of the southwestern willow flycatcher (Sogge et al. 2010). This subspecies may be able to coexist with cowbirds as a stable population in the absence of other threats (USFWS 2002). Brown-headed cowbirds appear to be more of a threat at small, isolated nesting sites (Sogge et al. 2010). A study in coastal central California showed that individuals nesting in less-dense vegetation with a more open canopy are more likely to be parasitized (Finch and Stoleson 2000). Thus, high-quality, dense riparian habitat is valuable not only because it provides suitable habitat but also because it may reduce the ability for cowbirds to parasitize southwestern willow flycatcher nests.

There is no information on possible competition between migrating *E. t. adastus* and *E. t. brewsteri* and nesting *extimus* in the Plan Area, although it is possible that the groups compete briefly for the same food resources.

Population Status and Trends

Global: Declining (NatureServe 2011)
State: Critically Imperiled (NatureServe 2011)
Within Plan Area: Likely Declining

From the mid-1900s to the 1980s, populations of southwestern willow flycatcher declined rapidly (Unitt 1987). As of 2007, there were 1,299 known territories occurring within 288 breeding sites throughout the southwestern willow flycatcher's range. Of the 1,299 territories, 930 were surveyed in 2007 and the remaining 369 had been surveyed in 2006 or earlier (Durst et al. 2008a). Short-term studies on southwestern willow flycatcher have shown either a decline in population or no trend (Finch and Stoleson 2000). Within the Plan Area, significant declines have occurred along the Lower Colorado River, and occupied sites have declined in the Mojave River (Durst et al. 2008a). Overall, this subspecies is considered to be in decline (NatureServe 2011).

The majority of known territories and breeding sites occur in Arizona, New Mexico, and California. As of 2007, 96 breeding sites supporting approximately 172 territories have been documented in California, accounting for about 33% of all documented breeding sites in the subspecies' range and 13% of all documented nesting territories for that year (Durst et al. 2008a). Arizona and New Mexico currently account for the majority of the documented breeding sites (57%) and documented territories (75%) (Durst et al. 2008a). In California, the largest populations are along the South Fork Kern River, the Owens River, San Luis Rey River, and Santa Margarita River (USFWS 2002); a portion of the Owens River occurs within the Plan Area (but few, if any, actual territories now occur within the Plan Area).

The other two California subspecies of willow flycatcher, *E. t. adastus* and *E. t. brewsteri*, have also suffered severe declines and consequently are also listed as endangered by the State of California. Intense agricultural and flood control activities in the Central Valley virtually eliminated the riparian habitat used by *E. t. brewsteri* (Serena 1982), and both *E. t. adastus* and *E. t. brewsteri* meadow habitats in the Sierra Nevada have been impacted by grazing (Stefani et al. 2001).

Threats and Environmental Stressors

The primary threat to the southwestern willow flycatcher is loss, modification, and fragmentation of suitable riparian habitat (Sogge et al. 2010). In general, increased human populations and development have resulted in a decline of riparian habitat, a habitat type that is naturally rare, patchy, and dynamic in the Southwest due to the varying hydrologic conditions of the region. The specific primary causes for loss and modification of riparian habitats have been dams and reservoirs, water diversion and groundwater pumping, channelization, flood control, agriculture, recreation, and urbanization (Sogge et al. 2010).

Impacts on suitable riparian habitat and conversion of adjacent native upland habitat have also resulted in indirect effects that are detrimental to this subspecies. Brown-headed cowbirds, discussed in the Ecological Relationships section above, are typically associated with anthropogenic influences, such as agriculture (cattle grazing), recreation (camp grounds and golf courses), and urbanization (lawns) (USFWS 2002). Although cowbird parasitism is not considered to be a primary threat to southwestern willow flycatcher, combined with other threats and stressors such as habitat loss and degradation, cowbird parasitism could be a significant contributor to population decline (USFWS 2002).

In California, the invasion of tamarisk and giant reed (*Arundo donax*) in riparian habitats has also been facilitated by anthropogenic disturbances (USFWS 2002). Although southwestern willow flycatcher is known to nest in monotypic stands of tamarisk, tamarisk is highly flammable and thereby has been suggested to pose a threat to southwestern willow flycatcher habitat (USFWS 2002; Finch and Stoleson 2000). However, while some territories have been lost in the last 20 years due to tamarisk fires, tamarisk has also supported many nesting territories, which have produced many hundreds of fledged flycatchers, which maintain and augment the population (Sogge, pers. comm. 2012). Additionally, Paxton et al. (2011b) concluded that using biocontrols such as tamarisk beetle (*Diorhabda* spp.) to eradicate tamarisk may negatively affect birds that have restricted distributions and sensitivity to seasonal defoliation, such as southwestern willow flycatcher, both in the short term and long term. Potential long term

adverse and beneficial effects will be related to the rate regeneration and/or restoration of cottonwood and willow riparian habitats relative to the rate of loss of tamarisk. Therefore, for southwestern willow flycatcher, its relationship to tamarisk is more complex than tamarisk simply increasing fire risk (Sogge, pers. comm. 2012).

Giant reed forms large monotypic stands that are unsuitable for the subspecies (USFWS 2002) and are also subject to large fires. The risk of fire has also increased along streams where the flow of water has been reduced, due to dams or flood control, allowing for the accumulation of fuel in the understory (USFWS 2002).

Grazing, cowbirds, and water removal (Owens Valley) projects continue to be a threat to Sierra Nevada populations of *E. t. brewsteri* and *E. t. adastus* within their breeding range. Within the Plan Area, the same threats mentioned above for *E. t. extimus* would affect *E. t. brewsteri* and *E. t. adastus* where they impact riparian migration corridors.

Conservation and Management Activities

Survey, monitoring, and research efforts increased significantly after the southwestern willow flycatcher was federally listed as endangered in 1995 (60 FR 10695–10715). Since then, statewide surveys have been initiated in Arizona, New Mexico, and Utah. Breeding and migration ecology, demography, and habitat research has been conducted in Arizona, New Mexico, and California (e.g., Crag and Williams 1998; Durst et al. 2008a, 2008b; Ellis et al. 2008; Hinojosa-Huerta et al. 2004; Langridge and Sogge 1997; Luff et al. 2000; Paxton et al. 2007; Sogge et al. 2010; Sogge and Paxton 2000). Range-wide population genetics work also has been conducted since the mid-1990s (USFWS 2002). Throughout the Southwest, several private, local, state, and regional efforts have formed in order to protect riparian habitats, including Partners in Flight and the Sonoran Bird Conservation Plan (USFWS 2002).

The Plan Area overlaps with the western part of the Lower Colorado River Recovery Unit, and the Basin and Mojave Recovery Unit identified in the recovery plan for southwestern willow flycatcher (USFWS 2002). The recovery plan sets forth alternative recovery criteria for the subspecies for downlisting to threatened and

additional criteria for delisting (USFWS 2002). One recovery criterion (Criterion A) for downlisting to threatened status is increasing the known total population to a minimum 1,950 territories that are geographically distributed to allow metapopulation function and which are maintained over a 5-year period. An alternative criterion (Criterion B) for downlisting the subspecies to threatened is to increase the population to a minimum of 1,500 territories that are geographically distributed among management units and recovery units, protect the habitat supporting willow flycatcher populations from threats and loss, and maintain the population for a minimum 3-year period. The criteria for delisting the southwestern willow flycatcher is achieving Criterion A, providing protection from threats and creating/securing enough habitat to ensure maintenance of the populations and habitats over time (USFWS 2002).

The recovery plan also describes actions to offset habitat impacts, mitigation efforts, and other conservation efforts undertaken to the point in time the recovery plan was published in 2002. These conservation efforts included the following:

- Annual cowbird trapping on Marine Corps Base, Camp Pendleton, beginning in 1983, and annual surveys and nest monitoring started in 1999.
- Cowbird trapping, habitat restoration, and other conservation efforts in the Prado Basin area of the Santa Ana River beginning in 1996.
- Cowbird trapping and flycatcher monitoring and research associated with the construction of Isabella Dam.
- Management activities to benefit the southwestern willow flycatcher associated with the Roosevelt Dam in Arizona, including habitat acquisition, fencing, restoration, cowbird trapping, research, and monitoring.
- Protection and management of the Audubon Kern River Preserve, California, and habitat in the Cliff-Gila Valley, New Mexico, by the Nature Conservancy.

Several habitat conservation plans that provide regulatory coverage for southwestern willow flycatcher have been implemented, including

the Clark County, Nevada, Habitat Conservation Plan; the Lower Colorado River Multi-Species Conservation Plan; the Western Riverside County Multiple Species Habitat Conservation Plan; the City and County of San Diego Multiple Species Conservation Programs; the San Diego Association of Governments North County Multiple Habitat Conservation Program; the Southern Orange County Habitat Conservation Plan; and the Sonoran Desert Multi-Species Conservation Plan. Each of these conservation plans provides for conservation/protection and management of riparian habitats that benefit southwestern willow flycatcher.

In 2005, the USFWS designated approximately 48,896 hectares (120,824 acres) of critical habitat for the southwestern willow flycatcher, including along the Mojave River in the Plan Area (70 FR 60886–61009). A proposed rule for revised critical habitat for the southwestern willow flycatcher was published in August 2011 (76 FR 50542-50629). Rather than designating aerial extent (i.e., total hectares) of critical habitat, as was done in the 2005 designation, the 2011 proposed rule expresses the total proposed critical habitat in terms of total stream length; approximately 3,364 stream kilometers (2,090 stream miles). The 2011 proposed rule designates the Mojave Management Unit, which includes a 35.7-kilometer (22.2-mile) segment of the Mojave River (which is substantially expanded downstream compared to the 2005 designation), a 11.2-kilometer (6.9-mile) segment of the West Fork Mojave River, a 19.6-kilometer (12.2-mile) segment of Holcomb Creek (outside the Plan Area), and a 20.0-kilometer (12.5-mile) segment of Deep Creek (which includes the Mojave River Forks Reservoir in the Plan Area, but most of which is outside the Plan Area). The proposed rule also designates the Amargosa Management Unit segments, which include a 12.3 kilometer (7.7 mile) segment of the Amargosa River and a 3.5-kilometer (2.2-mile) segment of Willow Creek (3.5 km, 2.2 mi) in Inyo and San Bernardino counties. Neither of these two segments is in the current 2005 critical habitat designation.

Although the current 2005 critical habitat designation (nor the 2011 proposed designation) does not require specific conservation measures, it requires that evaluations of potential impacts on critical habitat be made on projects with a federal nexus (e.g., a federal permit action or funding) and may result in protection measures to avoid

adverse modification or destruction of critical habitats associated with the project.

In 2010, the U.S. Geological Survey, in cooperation with the Bureau of Reclamation and the USFWS, developed a standardized survey protocol to be used for focused surveys throughout the range of the southwestern willow flycatcher (Sogge et al. 2010). This protocol provides information necessary to conduct and interpret survey results successfully, including a summary of basic ecological and population status information. Having a standardizing survey protocol allows for consistent data collection, reporting, and streamlined interpretation.

Restoration of breeding habitat for *E. t. adastus* and *E. t. brewsteri* has been a prime focus under the amended Sierra Nevada Forest Plan, and restoration efforts in the Owens Valley and near Mono Lake have improved breeding opportunities after original riparian nesting habitat was lost due to diversion of water to Los Angeles. All of these efforts are outside the Plan Area.

Data Characterization

At this time, information on the distribution and occurrence of the southwestern willow flycatcher within the Plan Area is limited, with very few documented occurrences. A greater level of confidence regarding the distribution of populations and isolated territories is needed in order understand the species' local status so that it can be managed adequately. Furthermore, the loss and degradation of riparian habitat is one of the most critical threats to the southwestern willow flycatcher. More information is needed regarding the distribution of suitable and potentially suitable habitat within the Plan Area and potential impacts that may be occurring in those areas, such as occupancy by invasive species and hydrologic alterations. As recovery efforts continue and the population size increases, an important question for recovery and management is the potential for geographic expansion of the subspecies' breeding range.

Further investigation on the wintering grounds for southwestern willow flycatcher is needed in order to ensure that this subspecies is being protected adequately. Additional studies on the boundaries of

the winter range and the quality of habitat used by this subspecies need to be conducted. Once this information is available, studies regarding the factors that limit survival of southwestern willow flycatcher during the winter can be conducted. Additionally, studies regarding threats to wintering grounds can be identified, followed by identification of methods needed, if any, to protect wintering grounds. Similar studies need to be conducted for migratory corridors used by this subspecies (Finch and Stoleson 2000). The same is true for *E. t. adastus* and *E. t. brewsteri*, especially in regard to how they use the Plan Area during annual migration periods.

Management and Monitoring Considerations

The recovery plan for the southwestern willow flycatcher outlines nine types of recovery actions: (1) increase and improve currently suitable and potentially suitable habitat; (2) increase metapopulation stability; (3) improve demographic parameters; (4) minimize threats to wintering and migration habitat; (5) survey and monitor; (6) conduct research; (7) provide public education and outreach; (8) assure implementation of laws, policies, and agreements that benefit the flycatcher; and (9) track recovery progress (USFWS 2002). As noted above, the Plan Area overlaps with portions of the Lower Colorado River Recovery Unit (Western Part) and the Basin and Mojave Recovery Unit. In the portion of the Lower Colorado River Recovery Unit overlapping the Plan Area, southwestern willow flycatcher occurrences are known from several locations south of Hoover Dam to the U.S.–Mexico border. In 2007, southwestern willow flycatcher territories were reported from the Hoover–Parker management unit and the Parker–Southern International Border management unit (Durst et al. 2008a). In the portion of the Basin and Mojave Recovery Unit overlapping the Plan Area, southwestern willow flycatcher occurrences are known from the Mojave River in the Victorville area. In 2007, southwestern willow flycatcher territories were reported from the Owens Management Unit, Amargosa Management Unit, Mojave Management Unit, and the Salton Management Unit (Durst et al. 2008a).

Given the apparent limited occurrence of the southwestern willow flycatcher in the Plan Area, management for the subspecies should

focus on removing existing or potential threats to riparian habitats, including invasive species, hydrologic changes in groundwater and surface water, and runoff from agriculture and urban uses. As discussed in Threats and Environmental Stressors, even though tamarisk is an invasive species, and ideally it would be eradicated and replaced with native cottonwood and willow habitats, tamarisk currently provides important nesting habitat for southwestern willow flycatcher (e.g., Paxton et al. 2011b; Shafroth et al. 2010). A temporal loss of tamarisk without available compensatory regeneration or restoration of native riparian habitat could have a substantial adverse effect on breeding southwestern willow flycatchers (e.g., Paxton et al. 2011b). Ellis et al. (2008), for example, recommends that tamarisk-dominant habitat in Arizona occupied by southwestern willow flycatcher should not be considered.

Ongoing monitoring and surveying efforts should continue in the Plan Area along the lower Colorado River, Mojave River, and Amargosa River and Willow Creek in areas containing suitable habitat in association with range-wide monitoring.

In addition to short-term cowbird control practices, such as trapping, long-term management practices may be needed for control of cowbird populations in southwestern willow flycatcher habitat if monitoring demonstrates that cowbirds are having significant local effects on southwestern willow flycatchers. Long-term management should emphasize reducing conditions known to attract cowbirds to riparian habitats, such as anthropogenic influences including golf courses, horse stables, and agricultural fields (Finch and Stoleson 2000; USFWS 2002). Providing educational programs for people residing near breeding populations would be beneficial in order to reduce anthropogenic conditions that attract cowbirds and domestic pets that can prey on birds.

Because southwestern willow flycatcher habitat also is threatened by catastrophic wildfires, especially in areas that support tamarisk (Finch and Stoleson 2000), specific fire management plans should be prepared in coordination with local firefighters for discrete occupied habitat areas.

All of the above management considerations relative to riparian habitats would also benefit *E. t. adastus* and *E. t. brewsteri* where they migrate through the Plan Area.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for willow flycatcher, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 329,611 acres of modeled suitable habitat for willow flycatcher in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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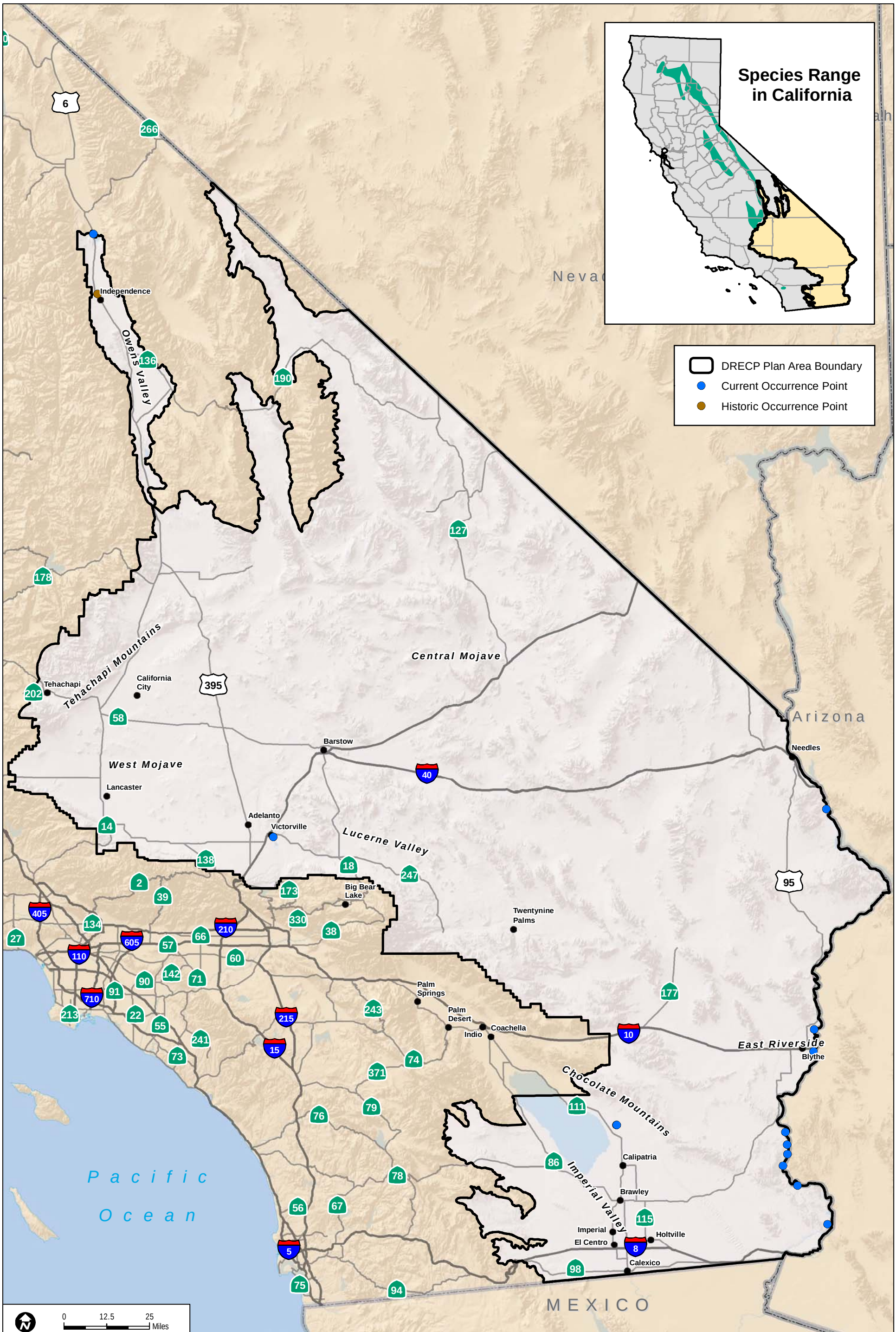


Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B11

Southwestern Willow Flycatcher Occurrences in the Plan Area

Yuma Clapper Rail
(*Rallus longirostris yumanensis*)

Legal Status

State: Threatened,
Fully Protected
Federal: Endangered
Critical Habitat: N/A
Recovery Planning: A federal recovery plan for the Yuma clapper rail was completed on February 4, 1983, by the U.S. Fish and Wildlife Service (USFWS 1983). A Draft Revised Recovery Plan was published in February 2010 (USFWS 2010).

Taxonomy

In 1902, Herbert Brown described a clapper rail he had captured near Yuma, Arizona, as a light-footed clapper rail (*Rallus levipes*). In 1923, Dickey described it as a new species, the Yuma clapper rail (*Rallus yumanensis*) (Todd 1986; USFWS 2010), based on several minor morphological differences from other clapper rails, as well as its isolated range and freshwater habitats (Banks and Tomlinson 1974).

Although there was some subsequent controversy over the rail's classification (Van Rossem 1929; Oberholser 1937), for over 60 years it has been widely treated as a subspecies of *R. longirostris* (i.e., *R. longirostris yumanensis*). This designation is consistent with available molecular genetic analysis (Fleischer et al. 1995).

A description of the species' physical characteristics can be found in the Draft Revised Recovery Plan (USFWS 2010).

Distribution

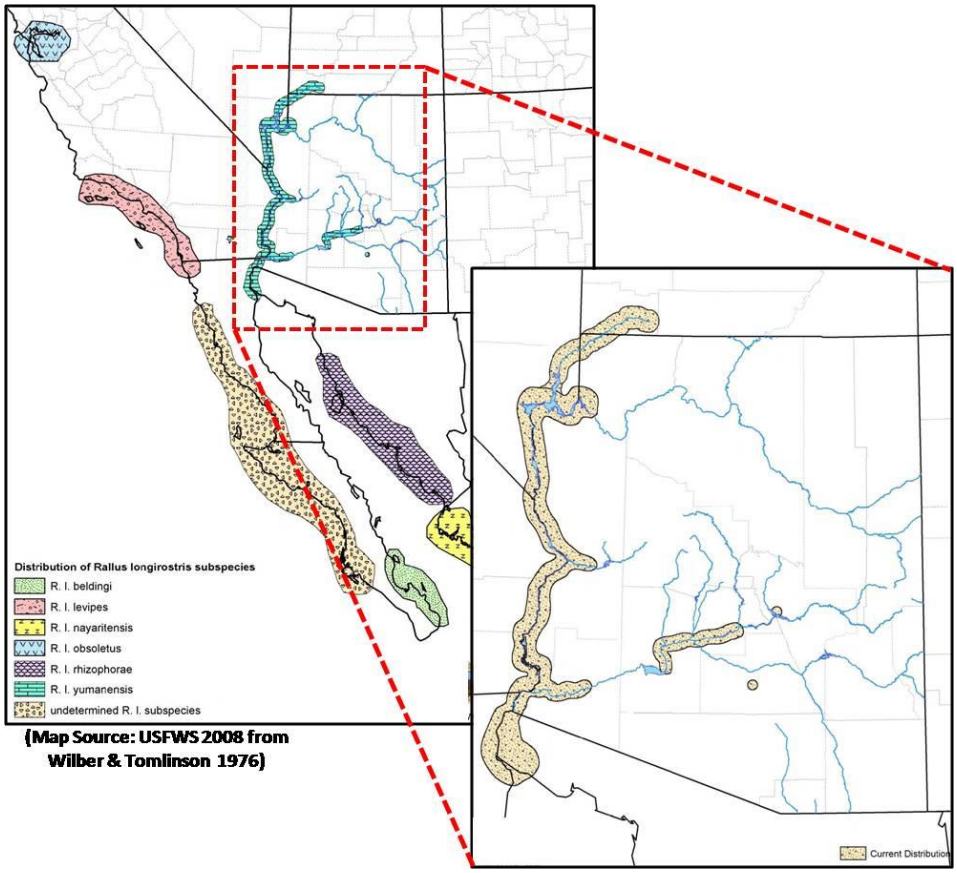
General

The Yuma clapper rail breeds along the lower Colorado River (including La Ciénega de Santa Clara in Mexico), the Gila River drainage in Arizona, Lake Mead (and the Overton Arm) and its local tributaries,

the Virgin River in Nevada and Utah, and the Salton Sea/Imperial Valley areas of California. Figure 1 shows the general breeding range of the species, while Figure SP-B16 indicates known occurrence in the Plan Area. In the Plan Area, the main habitat areas for this subspecies are located along the Colorado River and around the Salton Sea (including Dos Palmas Springs).

There are at least three "outlier" observations for Yuma clapper rail. In 1977, an individual was identified by vocalization on several days at Harper Lake northwest of Barstow (Figure SP-B16) but was not observed subsequently and was considered to be an unpaired individual (CDFW 2013). In 1978, the Yuma clapper rail was identified at Cronese Lake in the central Mojave (Garrett and Dunn 1981). In 1989, a single Yuma clapper rail was observed at the Ash Meadows National Wildlife Area located about 90 miles northwest of Las Vegas.

Figure 1. Range of the Yuma Clapper Rail



Distribution and Occurrences within the Plan Area

Historical

The historical distribution of the Yuma clapper rail is unclear. Todd (1986), in an extensive investigation of the Yuma clapper rail literature, reported that rails were first observed by J.G. Cooper near Fort Mojave in 1884. This is likely the earliest record. However, Joseph Grinnell performed an extensive survey of the Colorado River between Needles and Yuma in 1914 and did not record any observations of this species. However, he later documented the Yuma clapper rail from the lower Colorado River (Grinnell and Miller 1944, cited in Todd 1986). The Desert Renewable Energy Conservation Plan (DRECP) Area includes eight historical (i.e., pre-1990) records of the Yuma clapper rail in the California Natural Diversity Database (CNDDB) and others located just outside the Plan Area (Figure SP-B16) (CDFW 2013). Several of the historical occurrences occur along the lower Colorado River south of Parker to about 22 miles north of Yuma, Arizona (Figure SP-B16). Historical occurrences are also located at the Salton Sea, along the All American Canal, the New River, and the Holtville main drain in the Imperial Valley, as well as the single record each at Harper Lake in 1977, and Cronese Lake in 1978. (Figure SP-B16).

The Yuma clapper rail appears to respond positively to human activities that create habitat. Construction of dams both on the Colorado River and along adjacent tributaries has possibly contributed to the shift in the Yuma clapper rail's distribution (Ohmart and Smith 1973; Anderson and Ohmart 1985). Table 1 shows the relationship of upstream distribution of the Yuma clapper rail in relation to water management activities. These dams have the effect of creating sedimentation and backwater areas, thus providing additional shallow-water emergent habitat required by the Yuma clapper rail (CVCC 2007). Near the edge of the Salton Sea freshwater marsh ponds have been built and maintained to create habitat that now supports Yuma clapper rails.

Table 1. Upstream Distribution of the Yuma Clapper Rail and Relationship to Dam Construction and the Salton Sea Flood Event

Location	Year completed	Year Yuma clapper rail first found
Salton Sea	1905 (flooded)	1931
Laguna Dam	1905	1921
Headgate Dam	1941	1946
Parker Dam	1938	1954
Topock and Upper Lake Havasu	1938	1966
Needles Area	—	1982
Hoover Dam	1936	1986
Virgin River	—	1998

Source: USFWS 2010

Recent

The recent (i.e., since 1990) documented distribution of the Yuma clapper rail in the Plan Area is similar to the historic distribution, but with some apparent shift along the Colorado River. The distribution now ranges from about Lake Havasu to near Yuma, Arizona (Figure SP-B16). The recent distribution in the Salton Sea/Imperial Valley area is similar to the historic distribution. The Coachella Valley Habitat Conservation Plan (CVCC 2007) reports the Yuma Clapper Rail is found on Salt Creek and the Dos Palmas oasis in the southern Coachella Valley. The CNDDB contains 37 records for the period between 1990 and 2010 (CDFW 2013) and the USFWS database includes 20 records from 2004 to 2010 (USFWS 2011). The records from the USFWS database are located around the eastern edge of the Salton Sea, south of El Centro, and along the Colorado River near the Colorado River Indian Reservation and near the Imperial Reservoir. (It appears that there is some overlap between the USFWS and CNDDB databases for the period from 2004 to 2010, but the USFWS database contains the most recent data from USFWS protocol surveys.)

Yuma clapper rail has also colonized Ash Meadows National Wildlife Refuge (NWR) and has established a resident population there. Yuma clapper rail has also been known to inhabit Wixom Marsh near Seeley in the Imperial Valley. A May 2007 survey detected Yuma clapper rails defending breeding territories, and a Yuma clapper rail was heard calling in the marsh in January 2013. The marsh is thought to support two breeding territories.

Natural History

Habitat Requirements

Among the subspecies of clapper rail, only *yumanensis* is known to breed in freshwater marshes. By far, the preferred habitat consists of cattails (*Typha* spp.) and bulrush (*Scirpus* ssp.) (Anderson and Ohmart 1985; Todd 1986; Eddleman 1989). Eddleman (1989) found that habitat use by the subspecies on two study sites varied somewhat over different seasonal periods (i.e., early breeding, late breeding, post-breeding, early winter, and late winter), but that some combination of cattail and bulrush accounted for the majority of the observations across all periods. Combining data from the two study sites, use of cattail/bulrush habitats ranged from 66% of observations in the post-breeding period to 86% in the early breeding period (Eddleman 1989). Notably, on one of the sites, rails were observed in tamarisk (*Tamarix* spp.) second-most frequently behind cattail, with a range of 11% of the observations in the late winter period to 37% in the post-breeding and 36% in the early winter periods (Eddleman 1989). USFWS (2010) notes that the subspecies has been observed in shoreline areas with a mix of trees, including willow (*Salix* spp.) and tamarisk. However, although they are occasionally observed under the woody vegetation fringing a freshwater marsh, woody vegetation doesn't hold much habitat value for Yuma clapper rail compared to marsh vegetation (i.e., cattails and bulrushes).

Optimum habitat for the Yuma clapper rail results from a complex interplay of water levels, appropriate vegetation and vegetation characteristics (e.g., matting, dry areas, senescence), the timing of seasonal flooding, and possibly the timing of crayfish (*Procambarus clarkii* and *Orconectes virilis*, its primary prey) reproduction (Bennett and Ohmart 1978; Todd 1986). In a draft Recovery Plan

for the Yuma clapper rail, the USFWS (2010) characterized optimum habitat as consisting of:

> "… a mosaic of emergent vegetation averaging greater than 2 meters (6 feet) high (Anderson and Ohmart 1985; Eddleman 1989), shallow (less than 30 centimeters [12 inches]) open water areas either as channels or pools with minimal daily water fluctuation (Tomlinson and Todd 1973; Gould 1975), open dry ground (slightly higher than the water level) between water, vegetation, or marsh edge for foraging and movement (Gould 1975; Anderson and Ohmart 1985; Eddleman 1989; Conway et al. 1993), and a band of riparian vegetation on the higher ground along the fringes of the marsh that provides cover and buffer areas that may be used seasonally (Eddleman 1989)."

An overriding consideration for nesting by the Yuma clapper rail is that the nest substrate be stable (Eddleman 1989; USFWS 2006, 2010). Sparsely vegetated areas are more likely to be occupied if crayfish are abundant (Anderson and Ohmart, 1985). The Yuma clapper rail depends on a continuous source of water, most likely because crayfish are similarly dependent. However, the species also seems tolerant of seasonal fluctuations in water level that characterize the Colorado River (Eddleman 1989), as long as the change in level is not too abrupt (Conway and Eddleman 2000, cited in USFWS 2010). Similarly, Gould (1975) suggested that short-term changes in water level should be avoided. Rails may have several nests and can move eggs to nests that are less threatened if need be, but if the habitat dries out, rails will abandon the area (Bennett and Ohmart 1978; Johnson and Dinsmore 1985).

According to Gould (1975), in addition to the basic habitat requirements of standing water and marshland vegetation, the following habitat parameters are desirable to support high Yuma clapper rail densities:

1. "Water - flowing through many small channels, from 0.5 to 3 meters (1.5 to 10 feet) wide either covered by vegetation or

appearing as open water -or appearing as small bodies of open water, 0.02 to 0.2 hectare (0.05 to 0.5 acre) in size.

2. Extensive areas of water where depth is less 0.3 meter (1 foot). Little or no daily fluctuation in water level.
3. High ground found in strips, or less importantly as small isolated islands.
4. Emergent vegetation being cattail and bulrush with little or no carrizo cane [aka, giant reed (*Arundo donax*)]. In areas of carrizo cane, stem density is generally too high and there are few down stems."

An important aspect of Yuma clapper rail habitat is that over time, without occasional scouring by seasonal floods, marshes tend to become both overgrown (e.g., stem density too high), and much of the open or semi-open water fills with mats of old vegetation. The effects of this maturing process, or senescence, are that it becomes impossible for rails to move through vegetated habitat areas compared to open or semi-open aquatic habitat. Thus, foraging efficiency decreases as the habitat becomes choked with vegetation matting (Hinojosa-Huerta et al. 2008).

Foraging Requirements

As mentioned previously, the principal prey of the Yuma clapper rail are the two introduced species of crayfish that occur in the area (Inman et al. 1998). Ohmart and Tomlinson (1977) found that about 95% of the stomach contents of two Yuma clapper rail specimens were crayfish, leading them to suggest that the range shift of the Yuma clapper rail may have been facilitated by the introduction and spread of the crayfish. Other prey items taken by Yuma clapper rail include small fish, insects, amphibian larvae, clams, and other aquatic invertebrates (Todd 1986; USFWS 2010).

Reproduction

The Yuma clapper rail begins breeding activities in the early spring, usually in March or early April (Eddleman 1989), although mating calls may be heard as early as February (USFWS 2010). Breeding begins with the establishment of breeding territories. Birds occupying more

peripheral territories may mate a month or so later (Arizona Game and Fish Department 2007). Both males and females vigorously defend territories. Nesting occurs from March through May, but can vary with location and annual seasonal rainfall patterns (USFWS 2010).

Observed clutch sizes for 15 Yuma clapper rails nests in the lower Colorado River and Salton Sea ranged from 5 to 8 eggs (Eddleman and Conway 2012). Incubation was observed to last 23 to 28 days at nests in Arizona (Eddleman and Conway 2012). Both males and females incubate the eggs, with males incubating during the night shift and females incubating during the day (Eddleman 1989). Hatching success is high but juvenile mortality is also high (Bennett and Ohmart 1978; Eddleman 1989).

Young are precocial and within about 2 days of hatching they accompany adults on foraging trips, learning quickly to capture their own prey (Hunter et al. 1991). Family groups stay together for about 1 month, after which time the chicks separate from the parents. First flight occurs about 60 days after hatching (Arizona Game and Fish Department 2007).

Although nests may be from 6 centimeters (approximately 2.5 inches) to over 1 meter (approximately 3.3 feet) above the water level (average = 19.8 centimeters [approximately 7.8 inches]) (Eddleman 1989), as water levels rise, the birds may raise the level of existing nests or move eggs to a different nest. Consequently, the Yuma clapper rail may have several nests available for use (Conway and Eddleman 2000, cited in USFWS 2010).

Spatial Behavior

Migration and dispersal patterns of Yuma clapper rails are not well understood. The current scientific thinking is that Yuma clapper rails do not migrate seasonally. However, post breeding dispersal is likely possible over long distances. It was first assumed that the Yuma clapper rail migrated south during the winter (Smith 1974; Todd 1986), but Eddleman (1989) observed that up to 70% of the populations he studied remained at their site year-round in the lower Colorado River area. Also, as noted in Distribution and Occurrences, the observations for Yuma clapper rail at Harper Lake northwest of

Barstow in 1977 (CDFW 2013), another at Cronese Lake in 1978, an unpaired individual at Ash Meadows National Wildlife Area in 1989 (Garnett et al. 2004), and the finding in 2013 of an individual at a desert solar project located 32 miles from the nearest occupied habitat indicate that Yuma clapper rails are capable of long-distance movements. The purposes, frequency, and distances involved in long-range movements by Yuma clapper rails remain unclear, and is an important topic for future research (USFWS 2006, 2010).

The Yuma clapper rail also shows seasonal variability in its use of habitat and in its home range size (USFWS 2010). According to Eddleman (1989), there are five movement patterns by Yuma clapper rail outside of their breeding territory:

- Dispersal by juveniles
- Dispersal during the breeding season by unpaired males
- Movements of post-breeding adults
- Movements during late winter
- Home-range shifts associated with high water.

The triggers for these movements appear to be the need to find suitable habitat (juvenile dispersal, post-breeding movements, late winter movements), the need to find mates (late winter movements, movements of unpaired males during the breeding season), and/or the need to locate food (post-breeding and late winter movements) (Eddleman 1989).Home ranges are variable over different seasons, ranging on average from 7 to 8 hectares (17 to 20 acres) in the early and late breeding periods, to 15 hectares (37 acres) in the post-breeding period, and 24 hectares (59 acres) in the late winter period (Conway et al. 1993). Females have larger ranges than males in the post-breeding period at 21 hectares (51 acres), compared to 9 hectares (22 acres), but the two sexes have similar home range sizes the rest of the year (Eddleman 1989).

Ecological Relationships

The Yuma clapper rail is prey for several species, including coyote (*Canis latrans*), common raccoon (*Procyon lotor*), great horned owl (*Bubo virginianus*), Harris' hawk (*Parabuteo unicinctus*), and northern

harrier (*Circus cyaneus*) (USFWS 2010). Eddleman (1989) attributed 36 out of 37 known mortalities from natural causes to predation (50% by mammalian predators, 22% by avian predators, and 28% by unknown predators). Because these predators are generalists, however, the rail probably is not a critical element of their diets and likely is taken opportunistically.

As discussed previously, suitable habitat for the Yuma clapper rail depends on water levels, appropriate vegetation, the timing of seasonal flooding, and possibly the timing of crayfish reproduction. The subspecies appears to be particularly sensitive to water levels and may have several nests and can move eggs to nests that are less threatened by rising water levels if need be.

Population Status and Trends

Global: Vulnerable (NatureServe 2010)
State: Critically imperiled (NatureServe 2010)
Within Plan Area: Critically imperiled (NatureServe 2010)

The Yuma clapper rail in the United States has shown recent range extensions northward from the Colorado River Delta and the southern end of the Colorado River into Lake Mead and the Virgin River, indicating that the species is reproducing enough to support such a range shift (USFWS 2006, 2010). The species' first recovery plan (USFWS 1983) indicated that the breeding population had been stable for 10 years at the desired level of 700 to 1,000 individuals. As a result, a down-listing package was prepared for the Federal Register in 1983. However, subsequent flooding of important habitat on the lower Colorado River resulted in the proposal not being published (USFWS 2006).

The long-term assessment of population trends is complicated by several factors identified by the USFWS (2010), including:

- Inconsistencies in the proportion of suitable habitat surveyed in different years; and
- Different survey protocols, such as playback methods (e.g., continuous vs. intermittent call playback), seasons of surveys, and differing levels of surveyor experience.

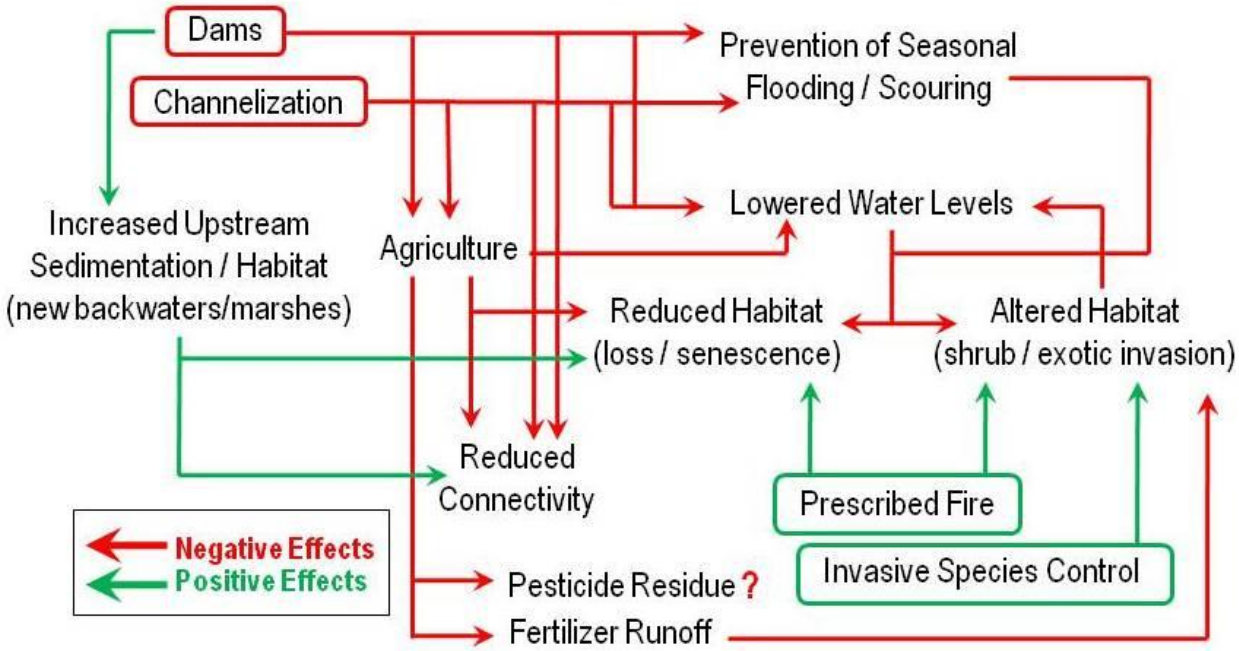
While the data for the United States populations of Yuma clapper rail do not allow for statistical population estimates, they do provide minimum number of rails in the census areas, which is the actual count of rails detected on survey routes, and which represents some subset of the actual population. Between 2000 and 2008, the minimum numbers in the United States ranged from 503 individuals in 2000 to 890 individuals in 2005 (USFWS 2010, Table 1). In the Plan Area, including the Colorado River and Salton Sea, the range over this same period was 472 individuals in 2001 to 849 individuals in 2005. The 2008 minimum number was 592 individuals along the Colorado River and at the Salton Sea (USFWS 2010). Within the lower Colorado River Delta region of Mexico (Ciénega de Santa Clara), Hinojosa-Huerta et al. (2008) documented a decline of 55% for the period of 1999 to 2002, but there was no statistically significant change between 1999 and 2006. The population was estimated to be 5,974 individuals (95% Confidence Interval = 4,698–7,482) in 2006, making it the largest documented population of the Yuma clapper rail.

Threats and Environmental Stressors

Habitat destruction and modification is the primary threat to the Yuma clapper rail (USFWS 2010). The natural hydrologic regime along the lower Colorado River has been altered by damming, channelization, and bank stabilization, the last of which has separated the main river channel from backwater and floodplain areas where marsh habitats would naturally form (USFWS 2010). While damming has likely created additional marsh habitat for rail in some areas, the dams have resulted in altered flood regimes from historical seasonal winter and spring flooding events that are necessary to maintain healthy marsh systems. These natural flooding events would have removed much of the thick matting of dead vegetation and build-up of sediments that allow for efficient foraging and escape from predation. Without active management, the value of these marsh habitats for Yuma clapper rail is reduced, and the habitat may disappear altogether (USFWS 2010). On the other hand, dams have also resulted in sedimentation of ancillary streams and creeks upstream, thereby increasing the extent of backwaters and marshes available for the Yuma clapper rail. This creation of new habitat has been cited as one reason for the shift of the species' range upstream (see Distribution and Occurrences within the Plan Area).

Figure 2 presents a generalized conceptual model of water management (dams, channelization) and their potential negative and beneficial effects on marsh habitat for the Yuma clapper rail.

Figure 2. A Generalized Conceptual Model for the Effects of Water Management on the Yuma Clapper Rail



Currently, the marshes at the Salton Sea Sonny Bono National Wildlife Refuge (NWR) and Imperial Wildlife Area are managed by flushing salts from the wetland ponds. Use of water for management of clapper rail habitat has increased since 2004 and may be constrained in the future by competing water uses, such as agriculture, that may increase the cost and availability of water (USFWS 2010).

Environmental contaminants may also pose threats to the species. Eddleman (1989) documented high levels of selenium in the Yuma clapper rail, its eggs, and its primary food source (i.e., crayfish). Similar levels of selenium were responsible for reproductive damage in mallards (*Anas platyrhynchos*) (Lemly and Smith 1987, cited in Eddleman 1989). Several studies have found high concentrations of selenium in the Colorado River and the Salton Sea (Andrews et al.

1997; King et al. 2000; Rusk 1991, cited in USFWS 2010). In the discussion of these studies, USFWS (2010) stated, "selenium levels in those studies were high enough to indicate the potential for exposure and adverse effects to Yuma clapper rails." Also, "… based on the available data, we do identify it [selenium] as a long-term threat to survival and recovery" (USFWS 2010, p. 16).

Conservation and Management Activities

A Yuma Clapper Rail Recovery Team was created in 1972 (USFWS 2006, 2010) that instituted survey protocols and additional research on the species. A formal recovery plan was created in 1983 and some of the recommended recovery actions commenced. Following these initial studies, the recovery team became inactive except for the coordination of annual surveys completed by volunteers from state and federal agencies (USFWS 2010).

In 1995, a group composed of local, state, and federal agencies; water and power agencies; environmental and recreational groups; and Native American tribes was formed to develop the Lower Colorado River Multi-Species Conservation Program (LCRMSCP). In December 2004, the LCRMSCP was completed (LCRMSCP 2004). Covering 26 species, including the Yuma clapper rail, the LCRMSCP calls for the creation of an additional 512 acres of Yuma clapper rail habitat and its management in an adaptive management framework to not only protect the Yuma clapper rail but also to understand how the management of threats and stressors affects Yuma clapper rail abundance.

Other programs to protect and enhance Yuma clapper rail habitat have been created at the Salton Sea by the U.S. Bureau of Reclamation (USFWS 2002) and at the Torres Martinez Desert Cahuilla Indians in 2005. Prescribed fire has been used to enhance Yuma clapper rail habitat at the Sonny Bono, Havasu, and Imperial NWRs, as well as the Mittry Lake Wildlife Area (USFWS 2010).

In 2006, a 5-year review of the recovery plan was completed (USFWS 2006), and the following five actions were recommended:

- Revise the recovery plan.
- Involve USFWS with the protection of the Ciénega de Santa Clara (Mexico), ensuring a continuous water source for this highly

significant sub-population, which, based on the 2006 population estimate by Hinojosa-Huerta et al. (2008) accounts for approximately 87% of the known Yuma clapper rail population.

- Establish new survey protocol and training using an adaptive management scenario to determine the effectiveness of management actions.
- Develop or revise management plans for the National Wildlife Refuges and State Wildlife Areas focusing on areas of declining Yuma clapper rail populations and habitat quality.
- Continue to support research efforts into the Yuma clapper rail, especially the possible effects of elevated selenium levels.

The federal government initiated efforts to implement these recommendations in 2007 (USFWS 2010). In February 2010, a Draft Revised Recovery Plan was released for public review (USFWS 2010). In this revision, the strategies used for the continued persistence of the Yuma clapper rail focused on "… providing long-term management and protection for a sufficient amount of core and other habitats to support a viable population of Yuma clapper rails, monitoring of populations and habitats, research to provide effective conservation and recovery, and application of research results and monitoring through adaptive management" (USFWS 2010, p. iv).

Data Characterization

Numerous surveys have been conducted for the Yuma clapper rail throughout its range in the U.S. and the Plan Area. Table 1 of the Draft Revised Recovery Plan shows that surveys were conducted along the lower Colorado River and at the Salton Sea every year from 1969 to 2007 (USFWS 2010). These data are not appropriate for estimating population sizes for various reasons, as discussed in Population Status and Trends, but they do provide information for the actual number of individuals observed along survey routes and allow some insight into occurrence population fluctuations and trends from year to year.

Despite the annual surveys and a reasonably good understanding of suitable habitat characteristics, information gaps that would inform management still exist, and research into the following topics should be conducted:

- The effects of elevated levels of selenium and pesticide residue on Yuma clapper rail reproduction and survival.
- The extent and importance of seasonal migration.
- Re-nesting.
- The effects of prescribed fire on senescent marshes and the Yuma clapper rail.
- The possible effects of increases in opportunistic predators associated with human presence and development (e.g., coyotes, feral pets, common raven (*Corvus corax*)).
- Genetic structure and gene flow.
- Seasonality and population structure of crayfish.
- The effects of human activities on the Yuma clapper rail, including noise, lighting, human presence, wildfire, and power lines.

Management and Monitoring Considerations

Management for the Yuma clapper rail should focus on maintaining high-quality marsh habitat. This includes not only the amount of available habitat, but the need for addressing water management issues important for maintaining high-habitat quality. Such issues include controlling water flows; establishing appropriate seasonal flooding and/or prescribed fire regimes to prevent decline and overgrowth of marshes; controlling of water levels during nesting periods; maintaining a habitat mosaic, that includes some upland areas; controlling exotic invasive species (e.g., tamarisk, giant reed); and controlling potentially harmful chemicals and other pollutants.

All management actions should be in concert with the goals of the Draft Revised Recovery Plan (USFWS 2010), which includes the following specific recommendations:

1. Define the minimum population size that must be maintained for the Yuma clapper rail in the U.S. to achieve recovery and document progress toward meeting that population size.
   - Determine the number of breeding birds in the U.S. that provides for a statistically and genetically secure population.
   - Conduct coordinated surveys for Yuma clapper rail in the U.S. to document when minimum viable population levels are met.
2. Define the physical parameters of and document the amount of Yuma clapper rail habitat in the U.S. needed to support the minimum viable population size.
   - Refine knowledge of rail use of habitats that support determination of the total amount of habitat needed in the U.S.
   - Develop techniques for managing habitats to maintain suitable conditions for Yuma clapper rail.
   - Complete an assessment of the amount and location of Yuma clapper rail habitat in the U.S. every 5 years.
3. Ensure that existing and new habitats for Yuma clapper rail are protected and managed for long-term habitat suitability.
   - Develop and implement management plans for all important federal- and state-owned core areas to maintain suitable habitat conditions.
   - Ensure all core areas in the U.S. have secure water sources that provide for a quantity and quality of water sufficient to manage existing and newly created rail habitat.
   - As possible, provide protection for other habitat areas supporting breeding Yuma clapper rails through management plans associated with easements, mitigation associated with federal actions, habitat conservation plans, safe harbor agreements, the Partners for Fish and Wildlife Program, tribal cooperation, and other options.

4. Provide a mechanism for coordination and implementation of recovery actions.
   - Establish a recovery implementation team with responsibilities for implementing recovery activities, with emphasis on tasks relating to survey management, research, and development of partnerships.
   - Cooperate with partners in Mexico on issues related to long-term survival of Yuma clapper rail.

Species Modeled Habitat Distribution

This section provides the results of habitat modeling for Yuma clapper rail, using available spatial information and occurrence information, as appropriate. For this reason, the term "modeled suitable habitat" is used in this section to distinguish modeled habitat from the habitat information provided in Habitat Requirements, which may include additional habitat and/or microhabitat factors that are important for species occupation, but for which information is not available for habitat modeling.

There are 54,978 acres of modeled suitable habitat for Yuma clapper rail in the Plan Area. Appendix C includes a figure showing the modeled suitable habitat in the Plan Area.

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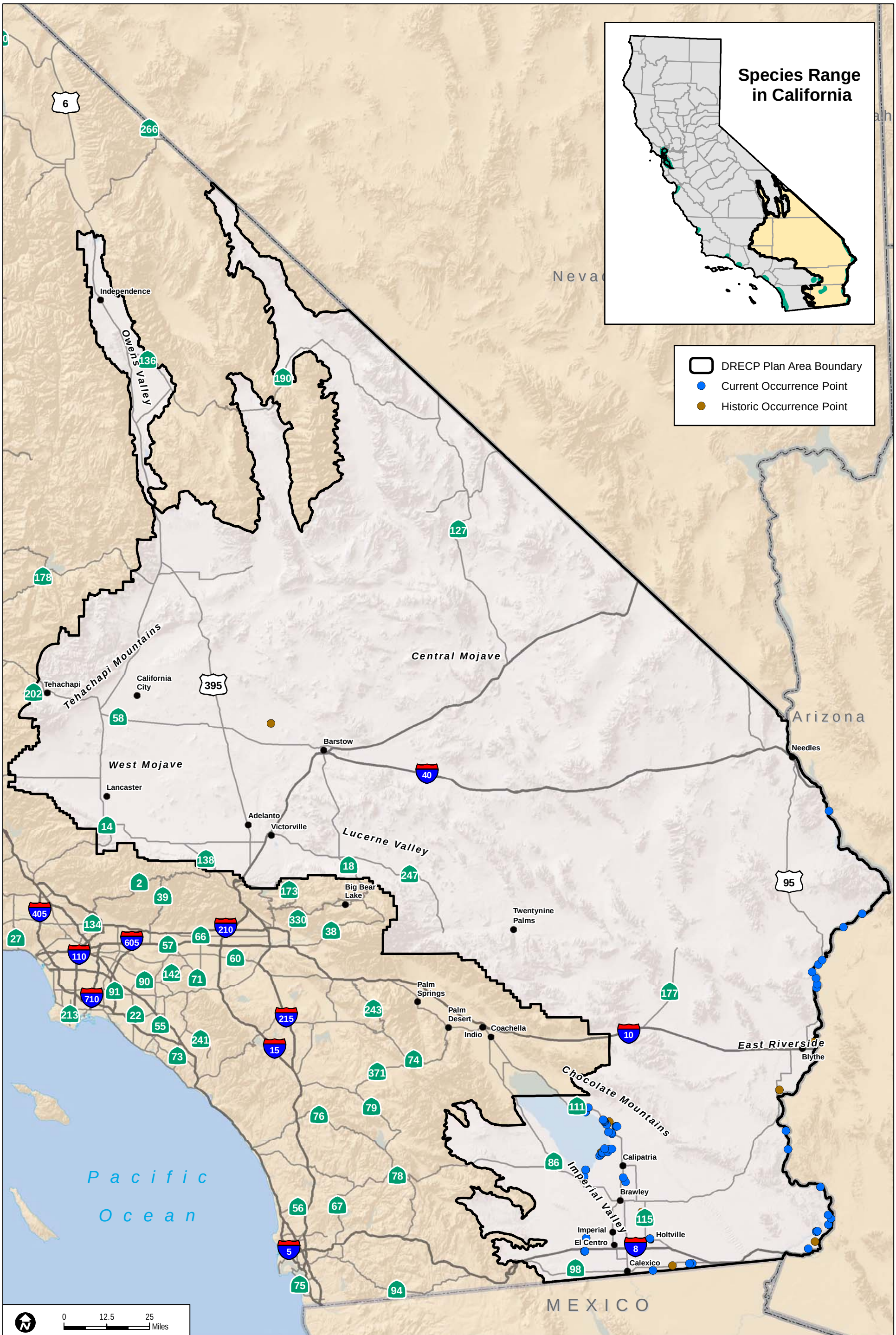

Sources: ESRI (2014); DRECP Species Occurrence Database (2013), CWHR (2008)

FIGURE SP-B14

Yuma Clapper Rail Occurrences in the Plan Area

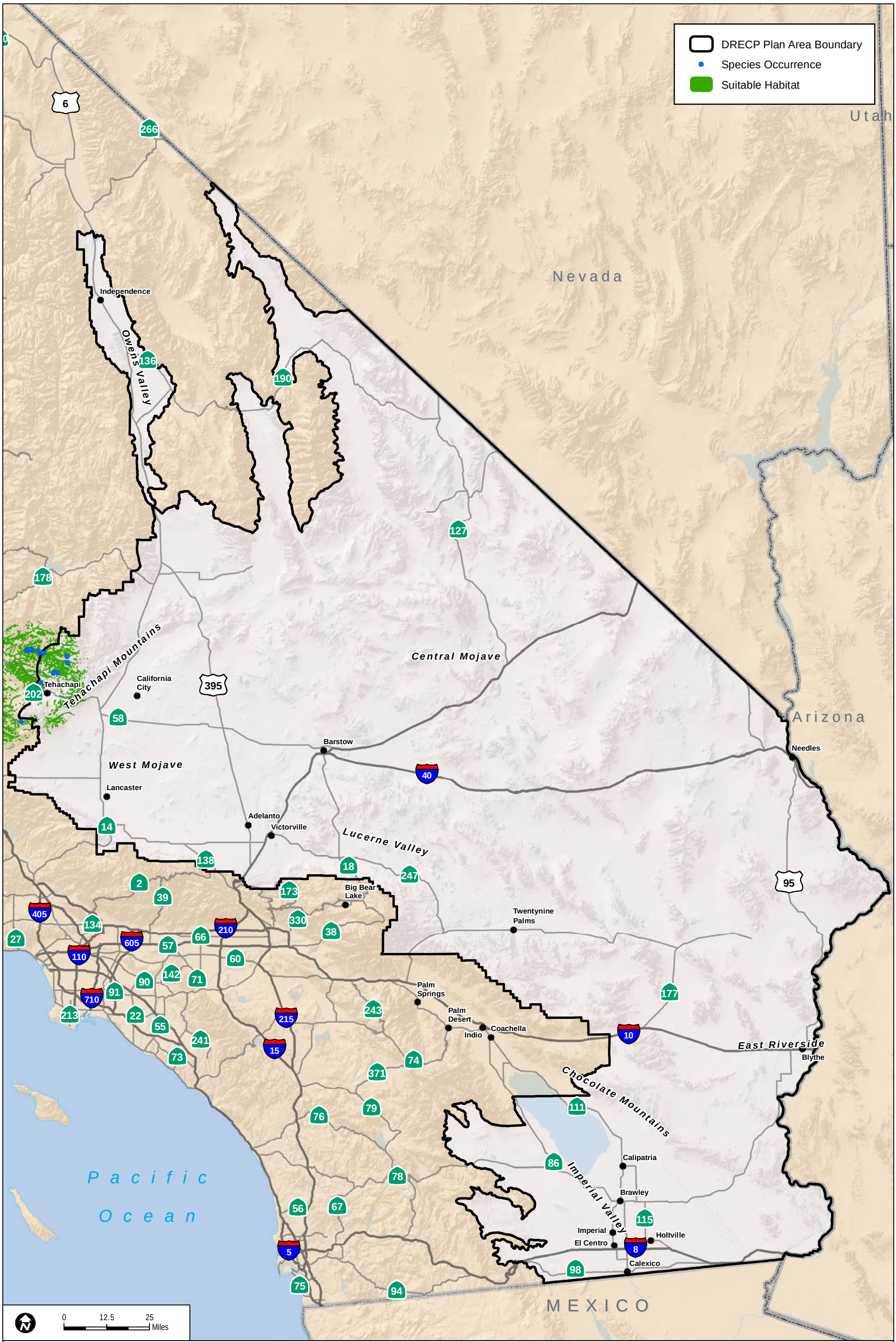

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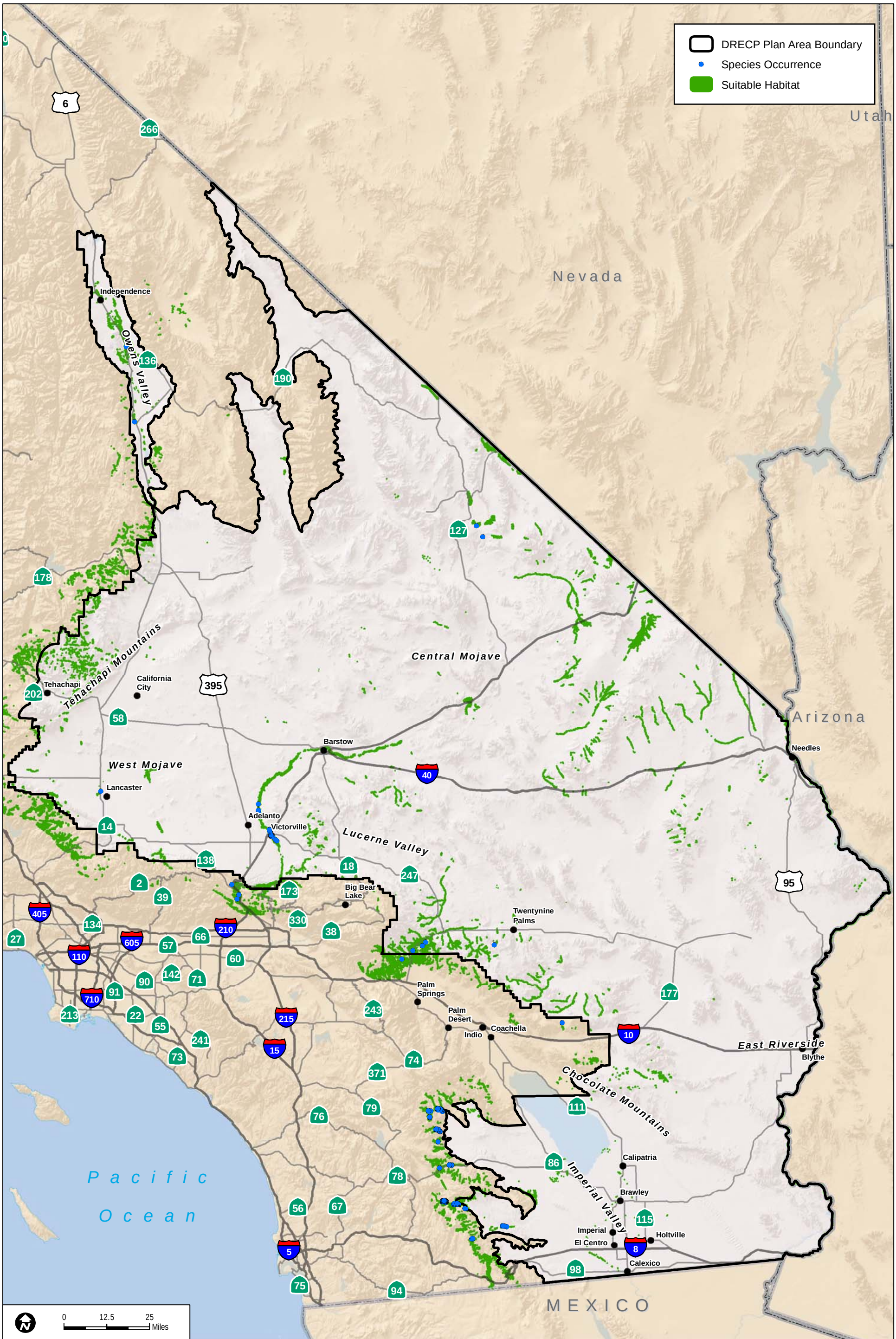
FIGURE SM-A01

Draft Species Habitat Model Results for Tehachapi Slender Salamander



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B01

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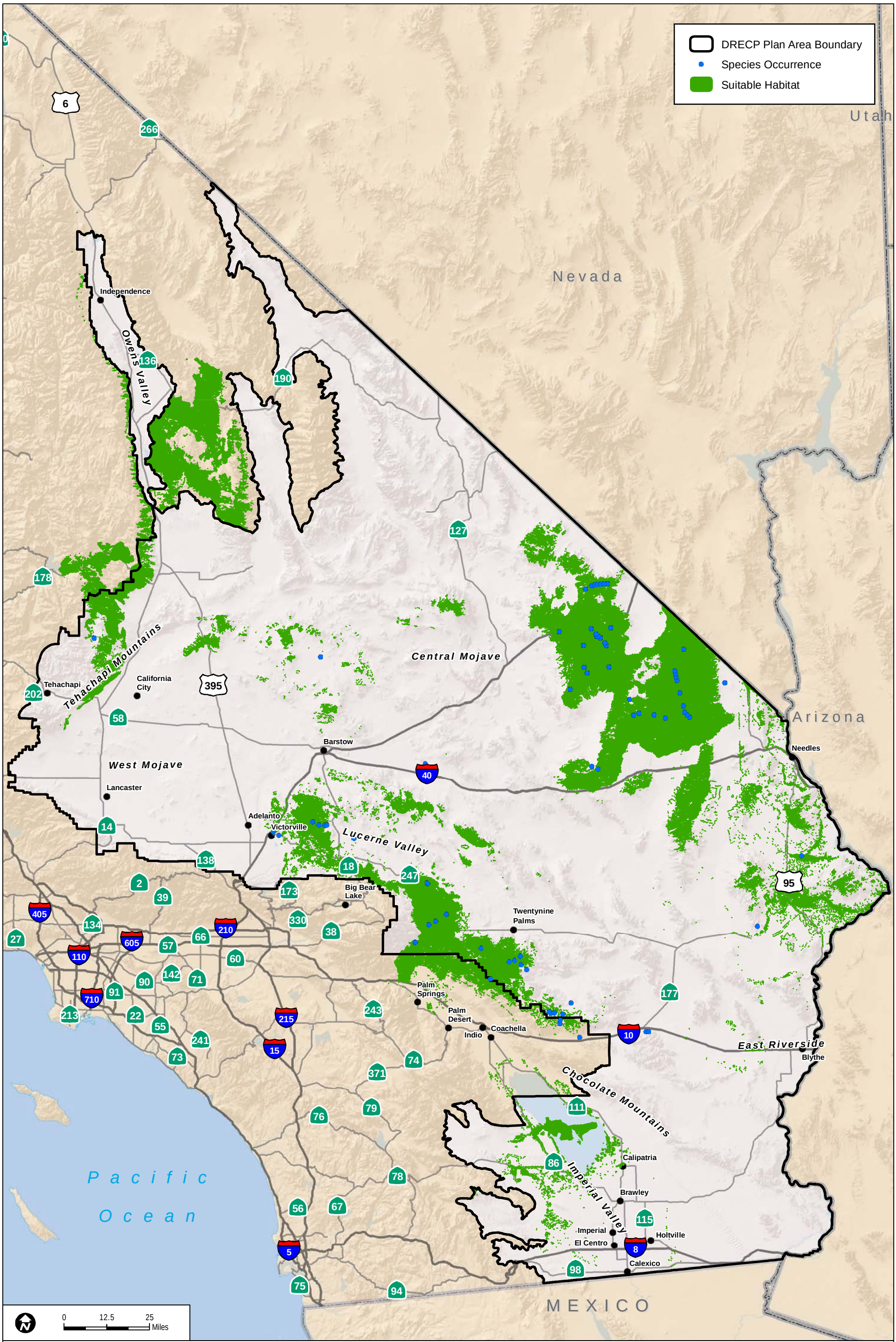

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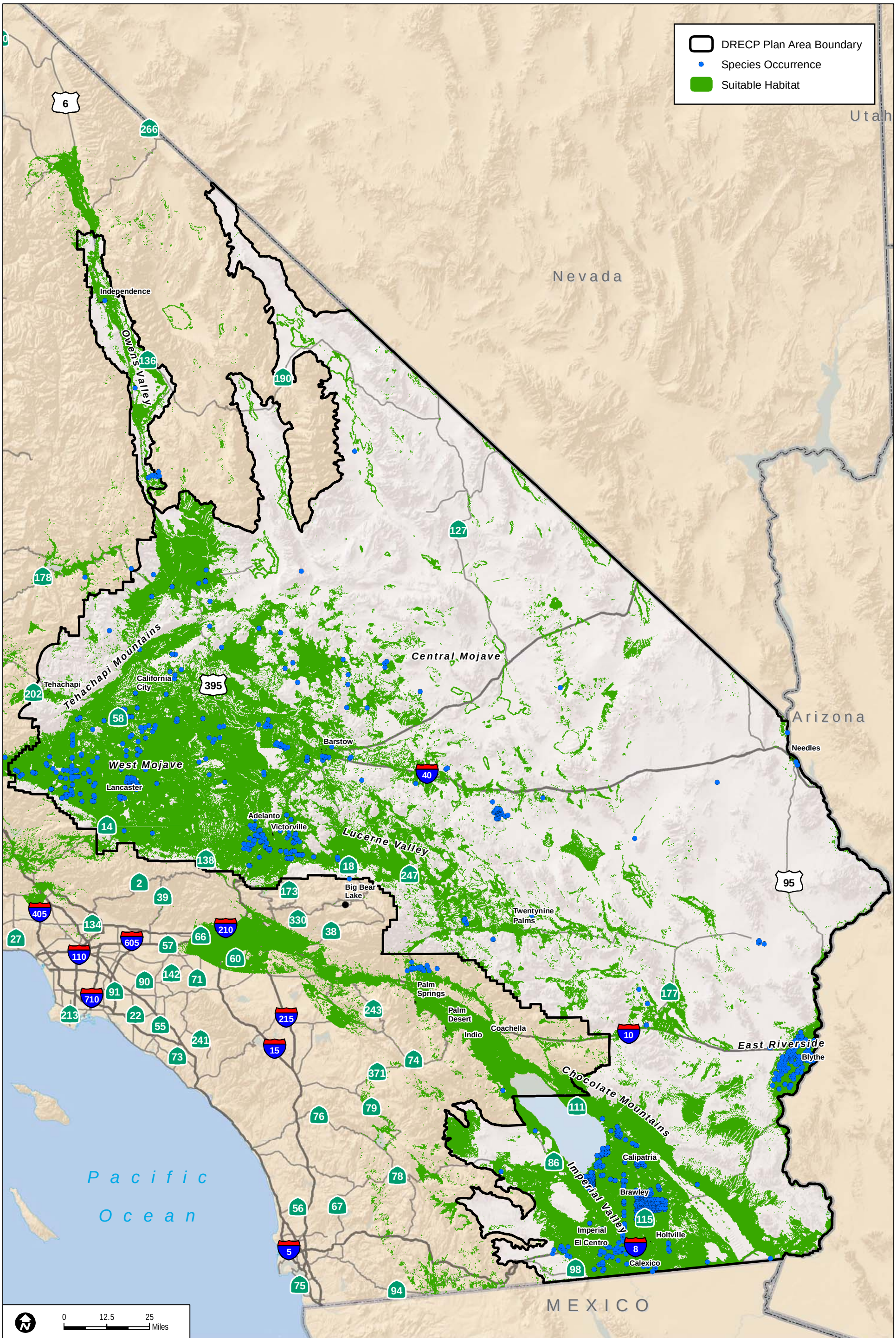
FIGURE SM-B02

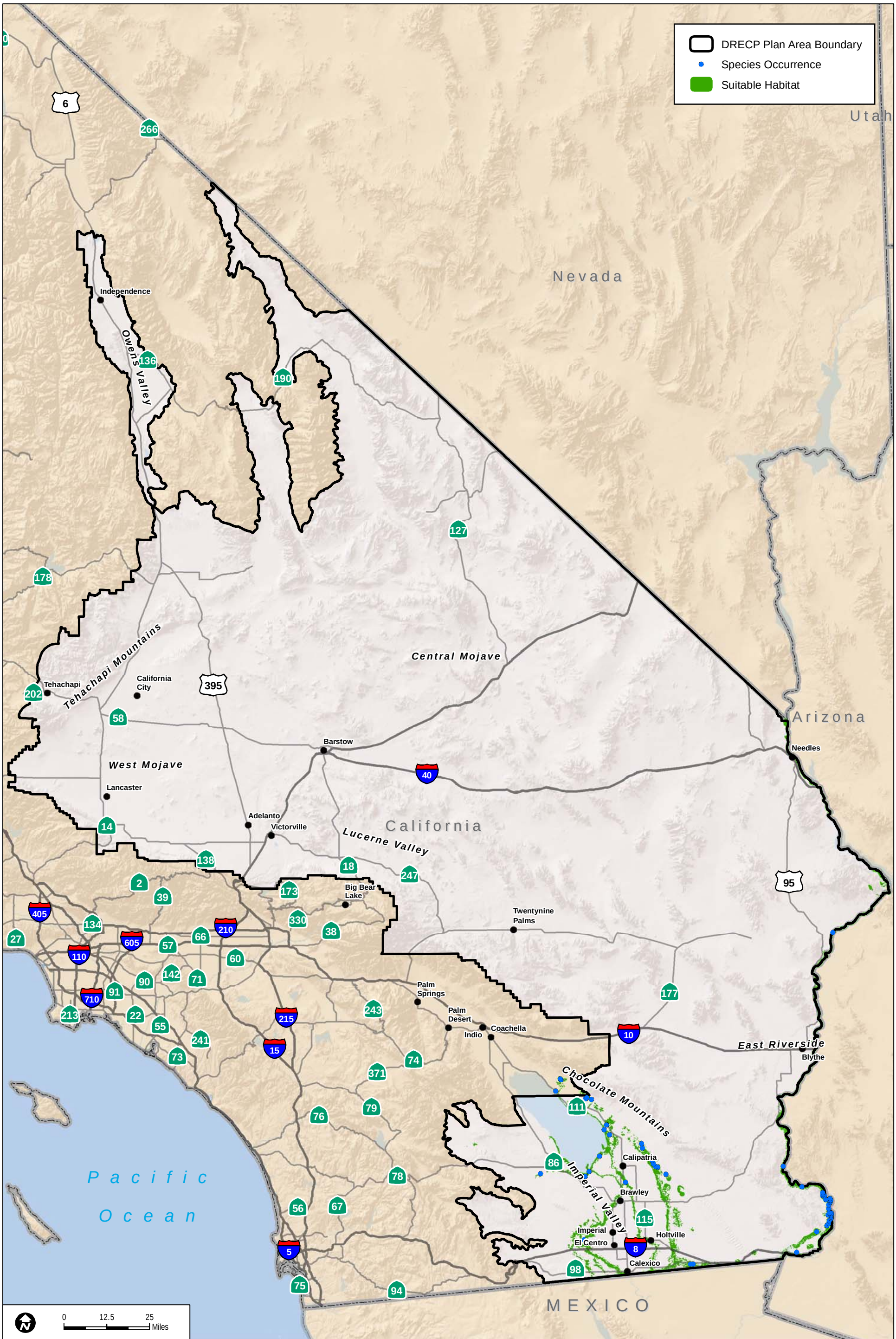
Draft Species Habitat Model Results for Bendire's Thrasher



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B03

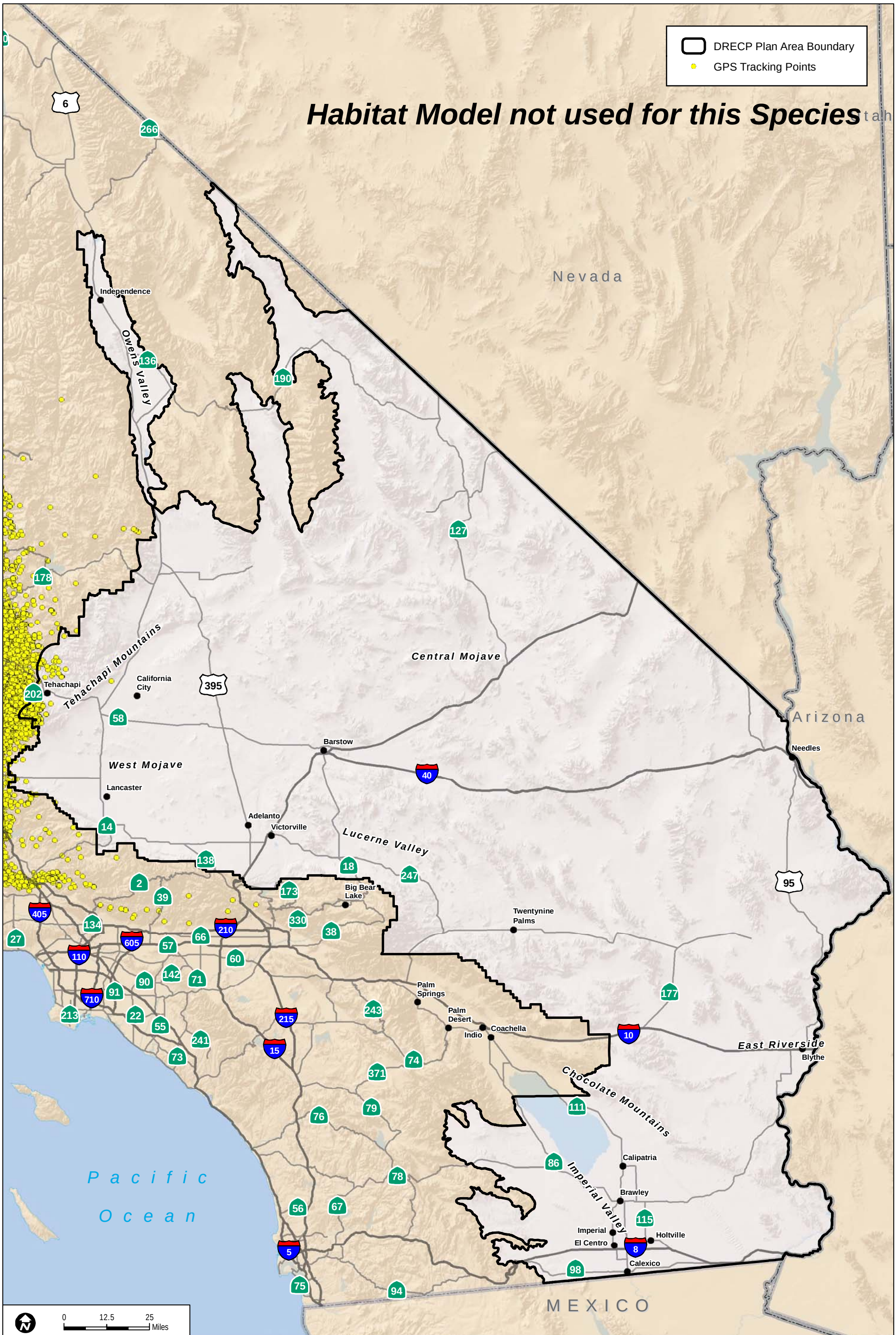
Draft Species Habitat Model Results for Burrowing Owl



Sources: ESRI (2014): DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B04

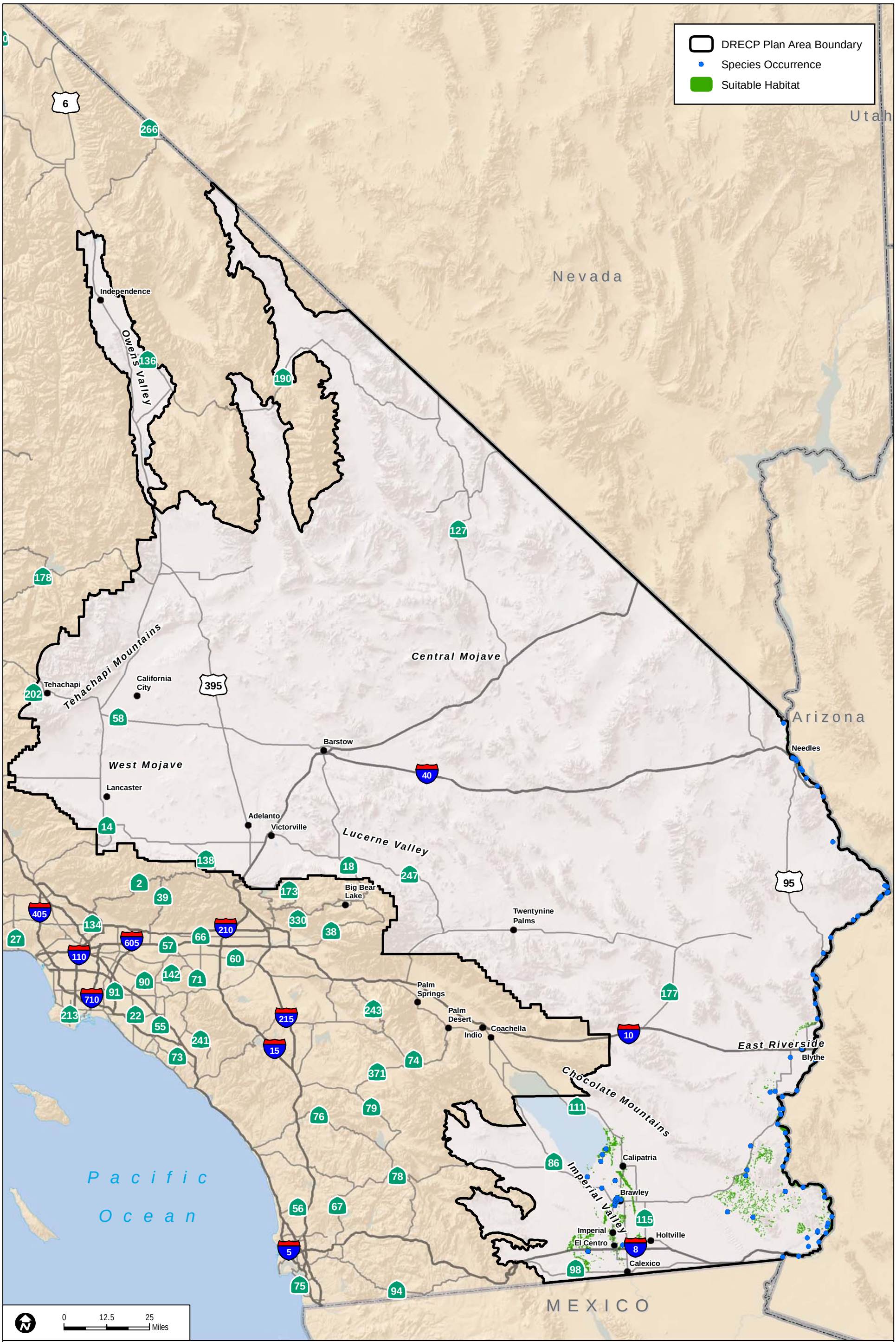
Draft Species Habitat Model Results for California Black Rail



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); USFWS (2013)

FIGURE SM-B05

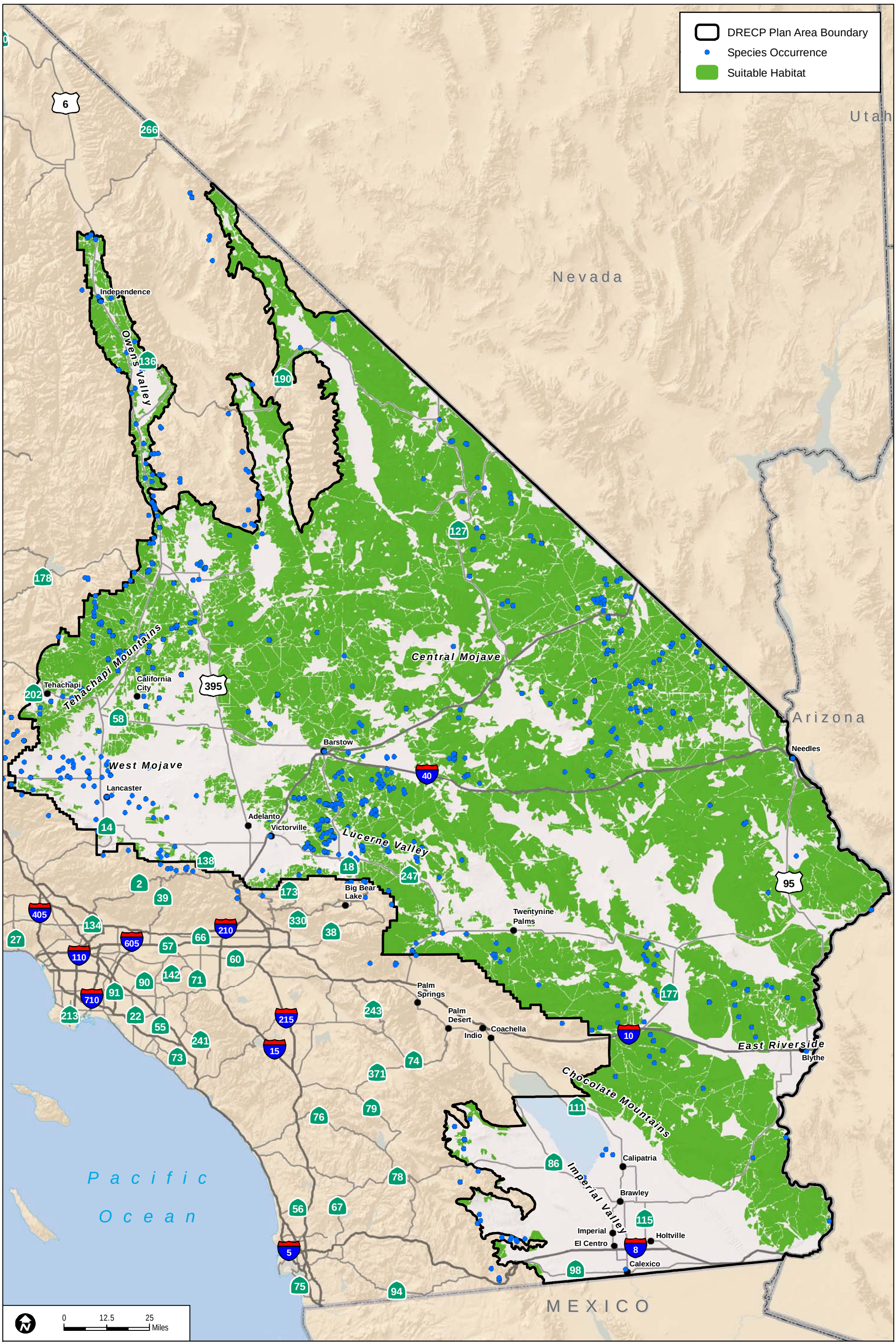
Draft Species Habitat Model Results for California Condor



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B06

Draft Species Habitat Model Results for Gila Woodpecker



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B07

Draft Species Habitat Model Results for Golden Eagle

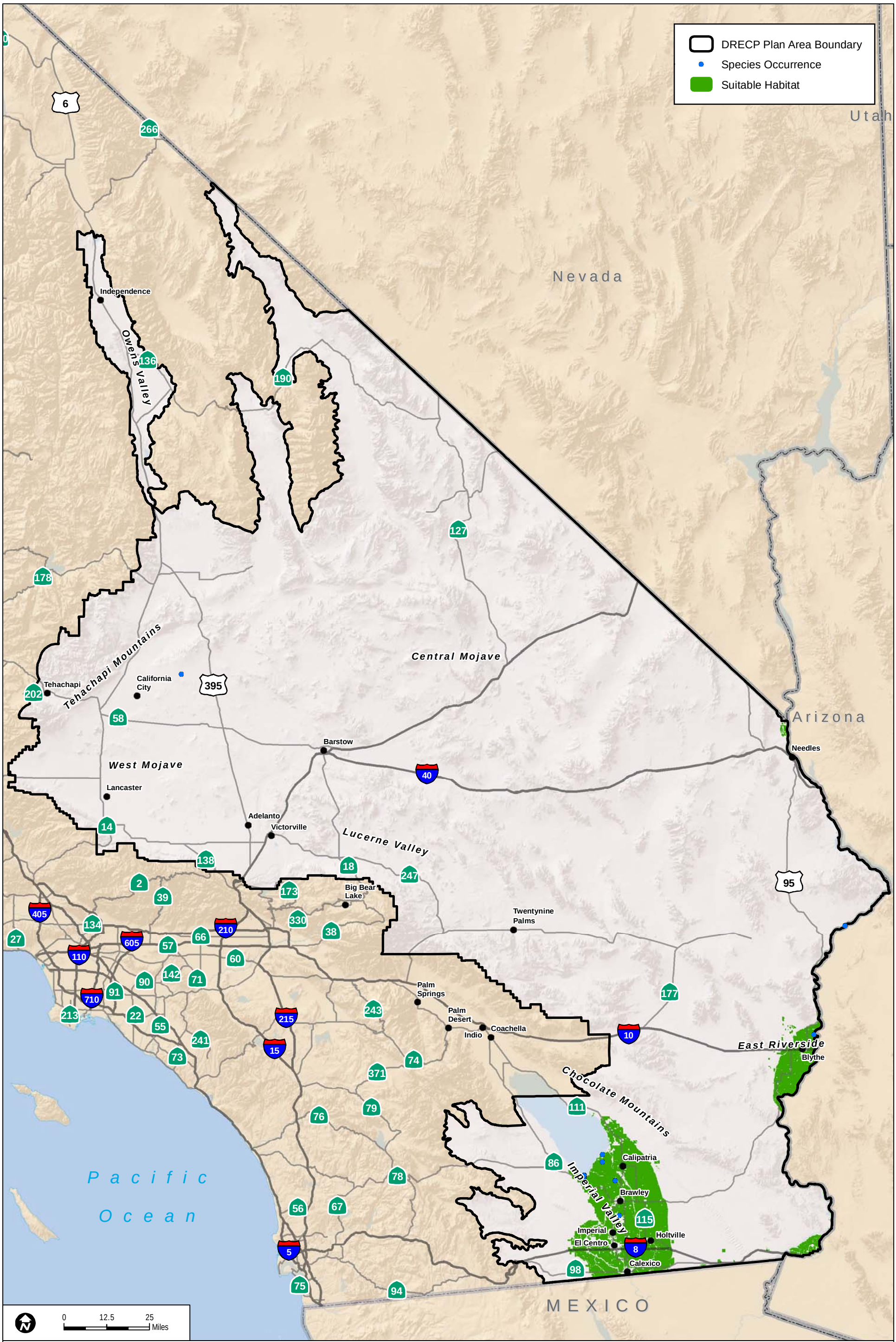


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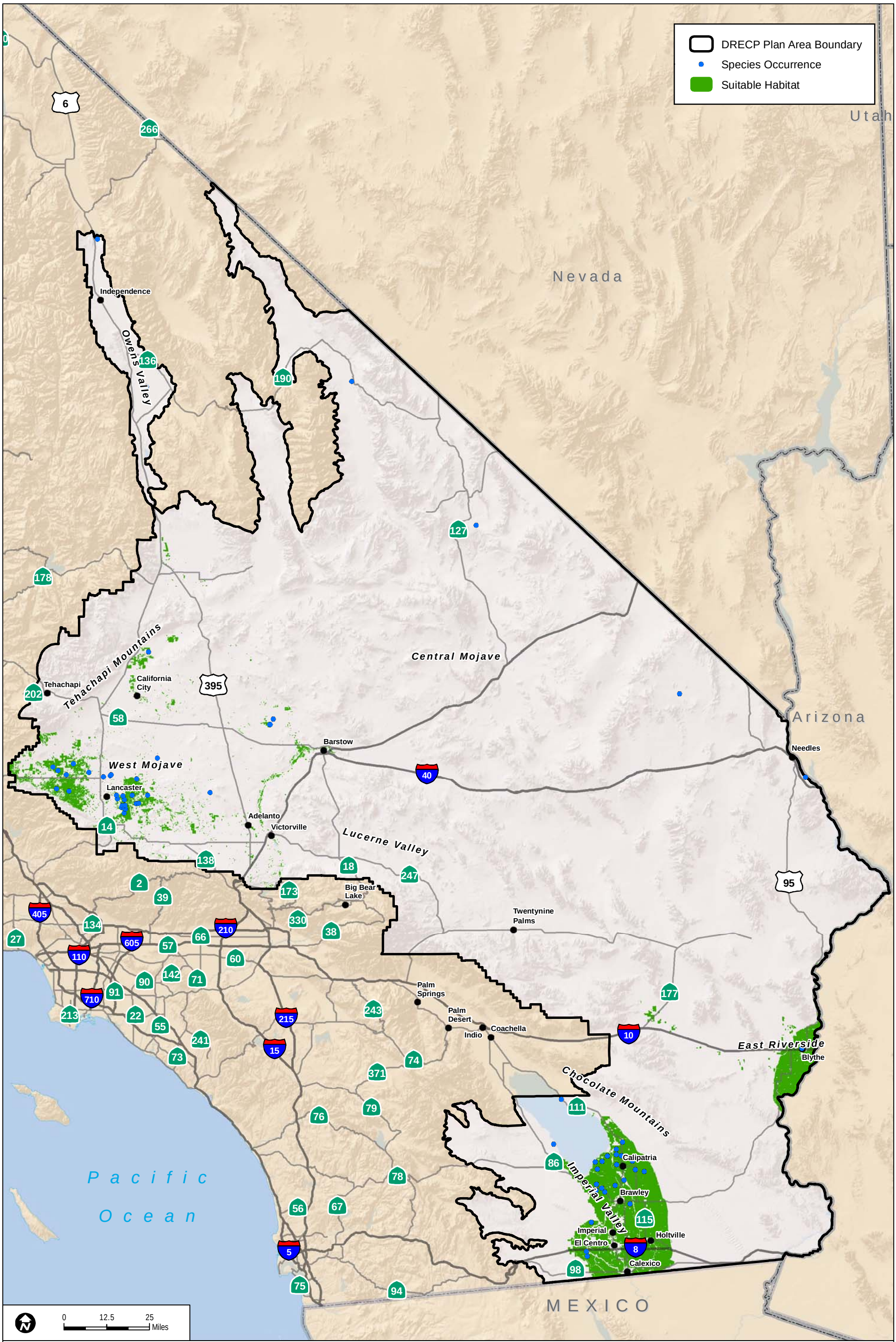
FIGURE SM-B08

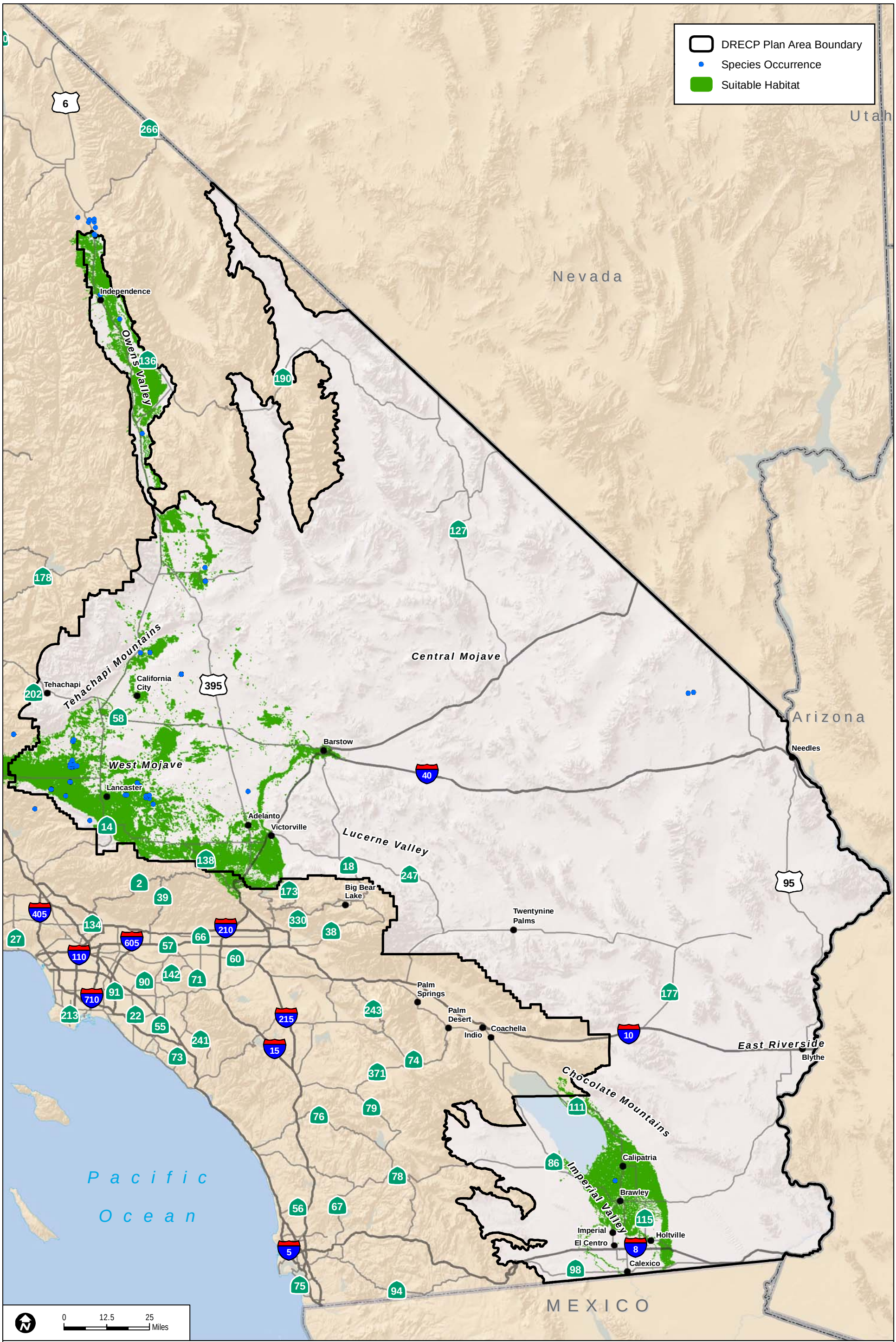
Draft Species Habitat Model Results for Greater Sandhill Crane



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B09

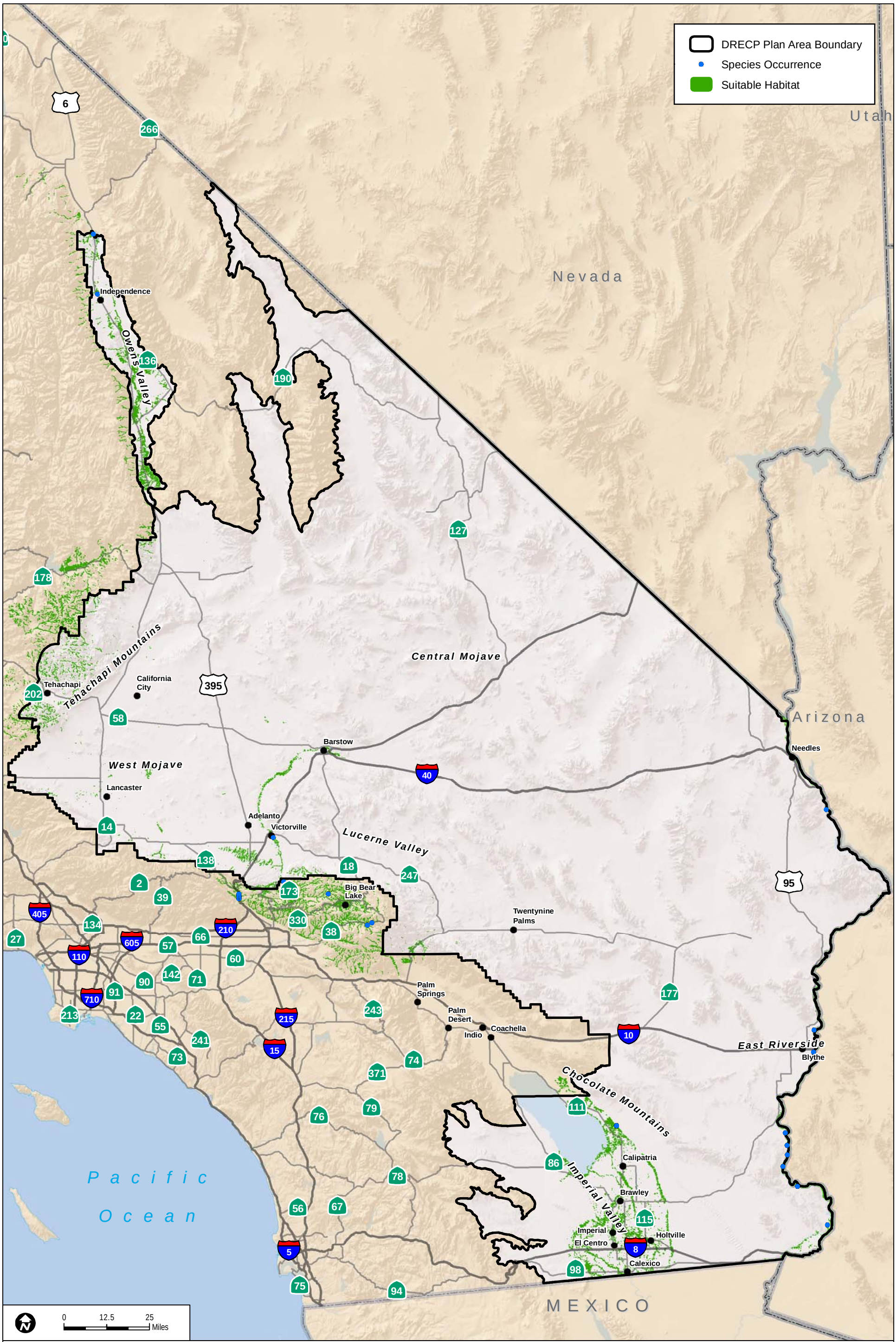
Draft Species Habitat Model Results for Mountain Plover



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B10

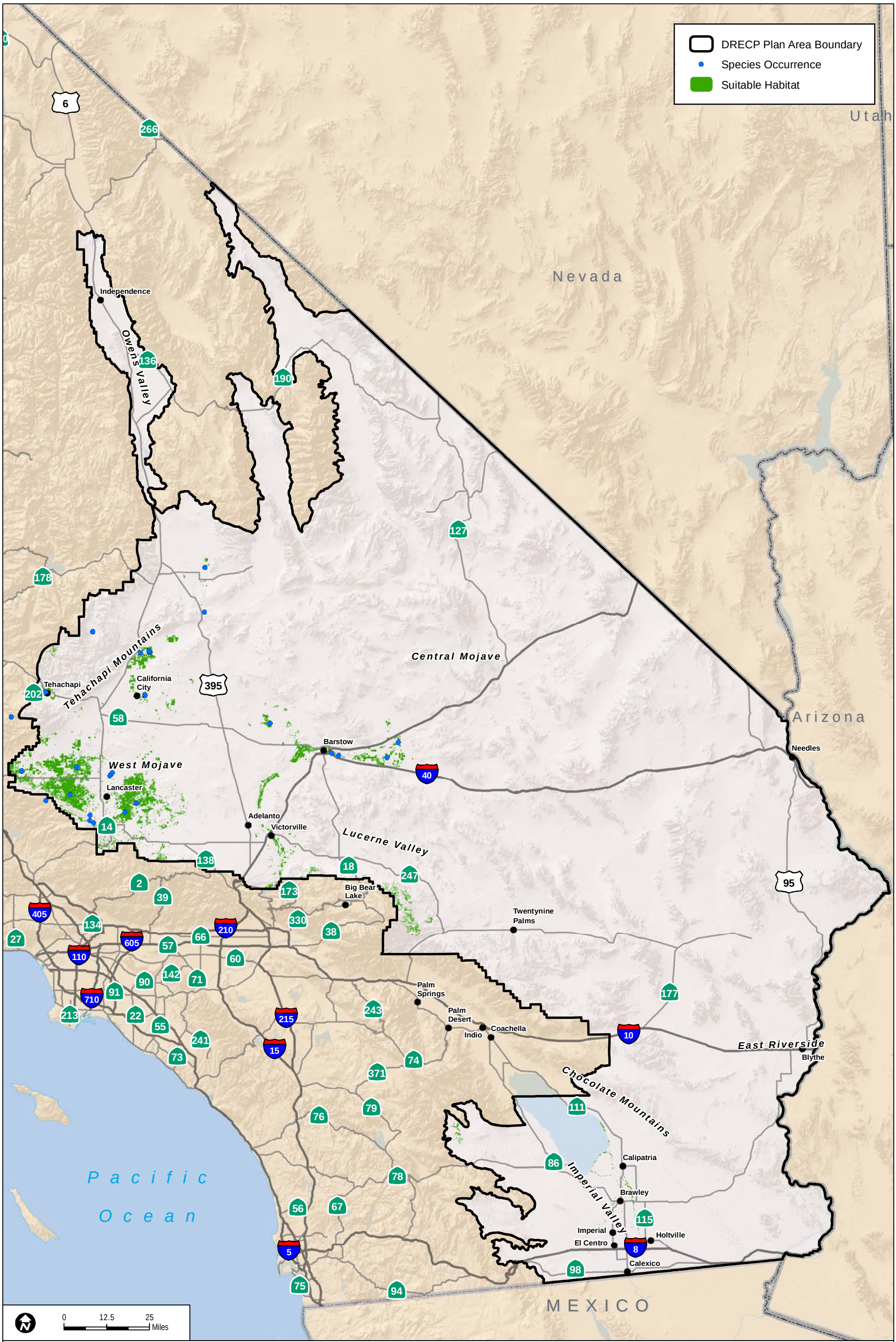
Draft Species Habitat Model Results for Swainson's Hawk



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B11

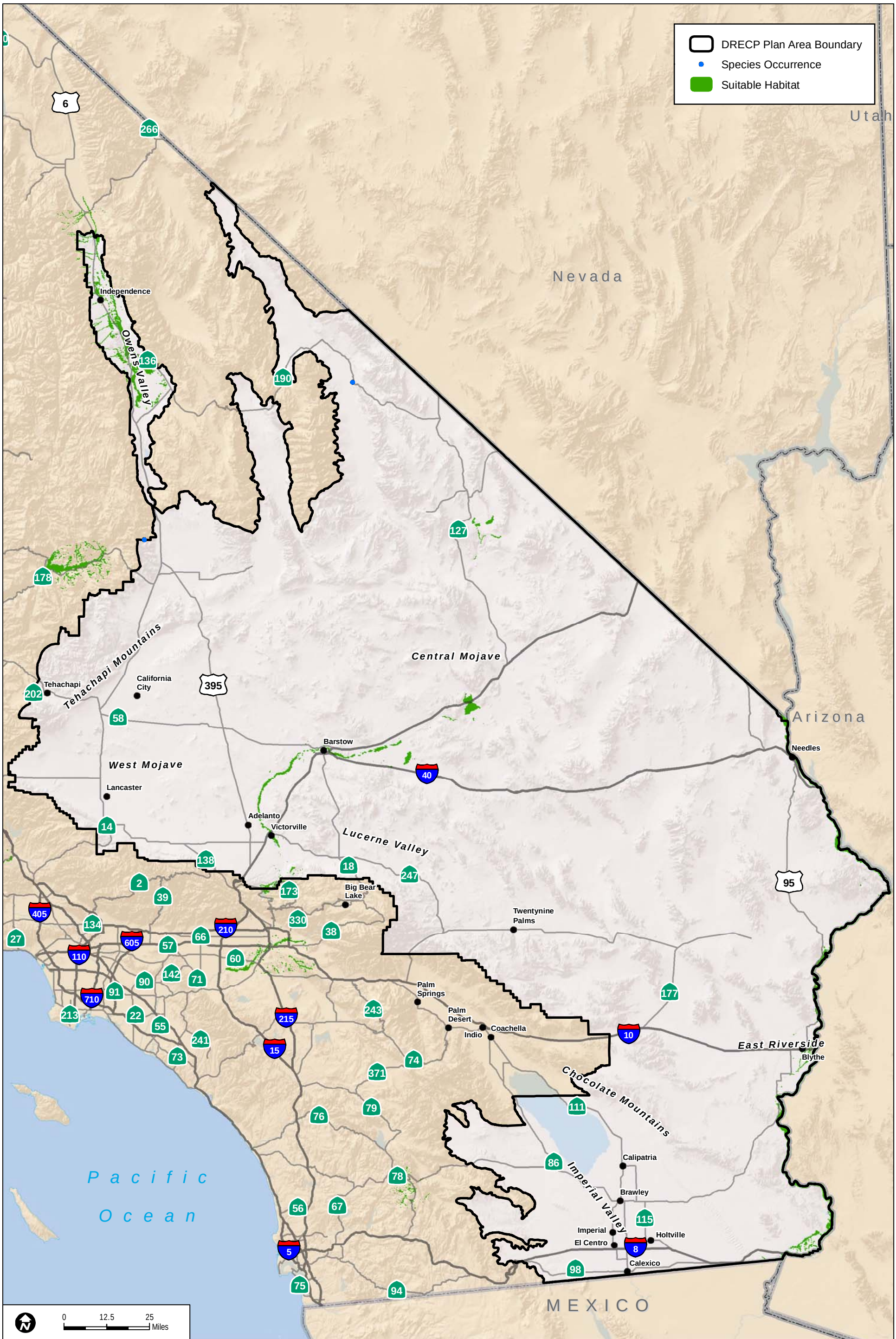
Draft Species Habitat Model Results for Southwestern Willow Flycatcher



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B12

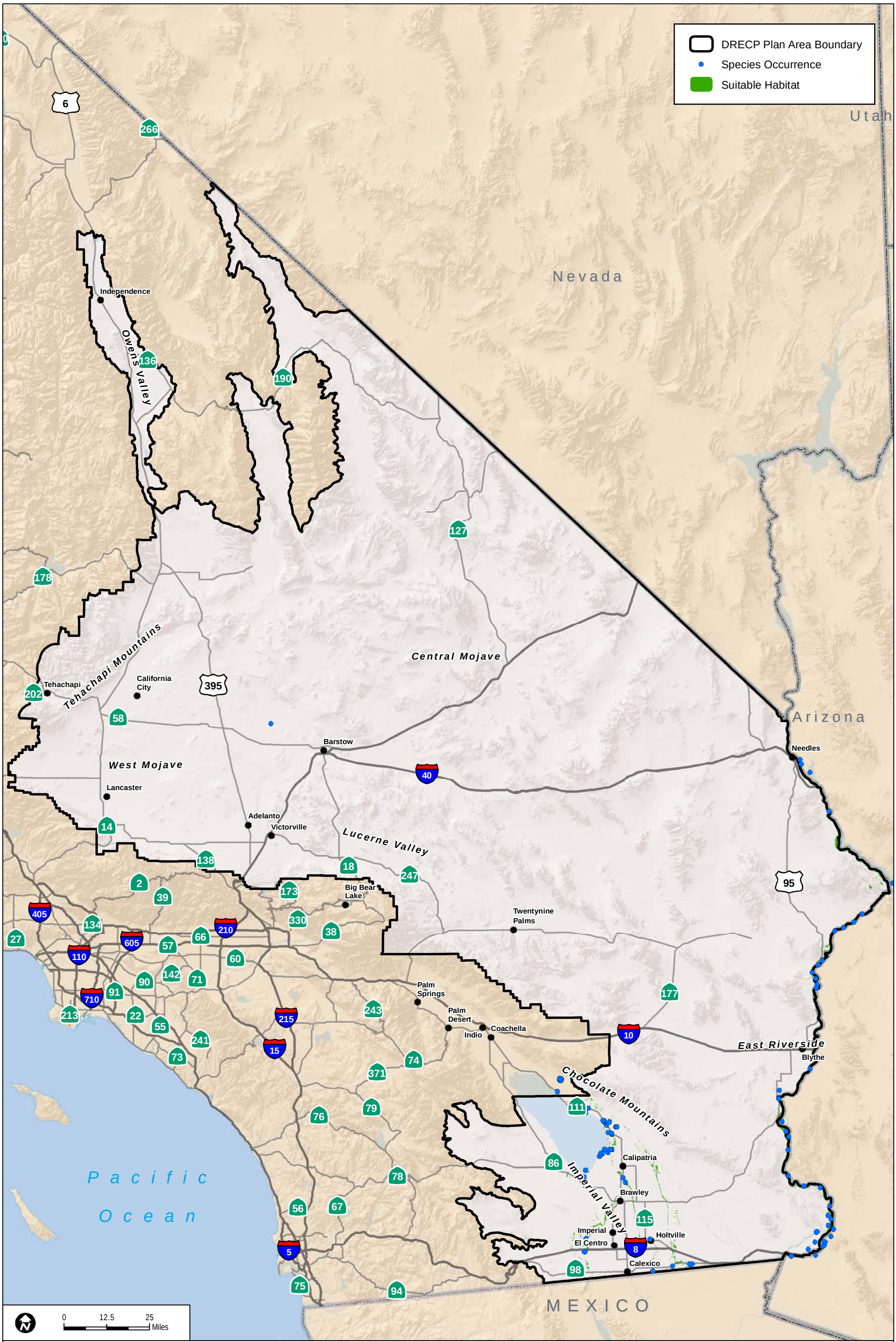
Draft Species Habitat Model Results for Tricolored Blackbird



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B13

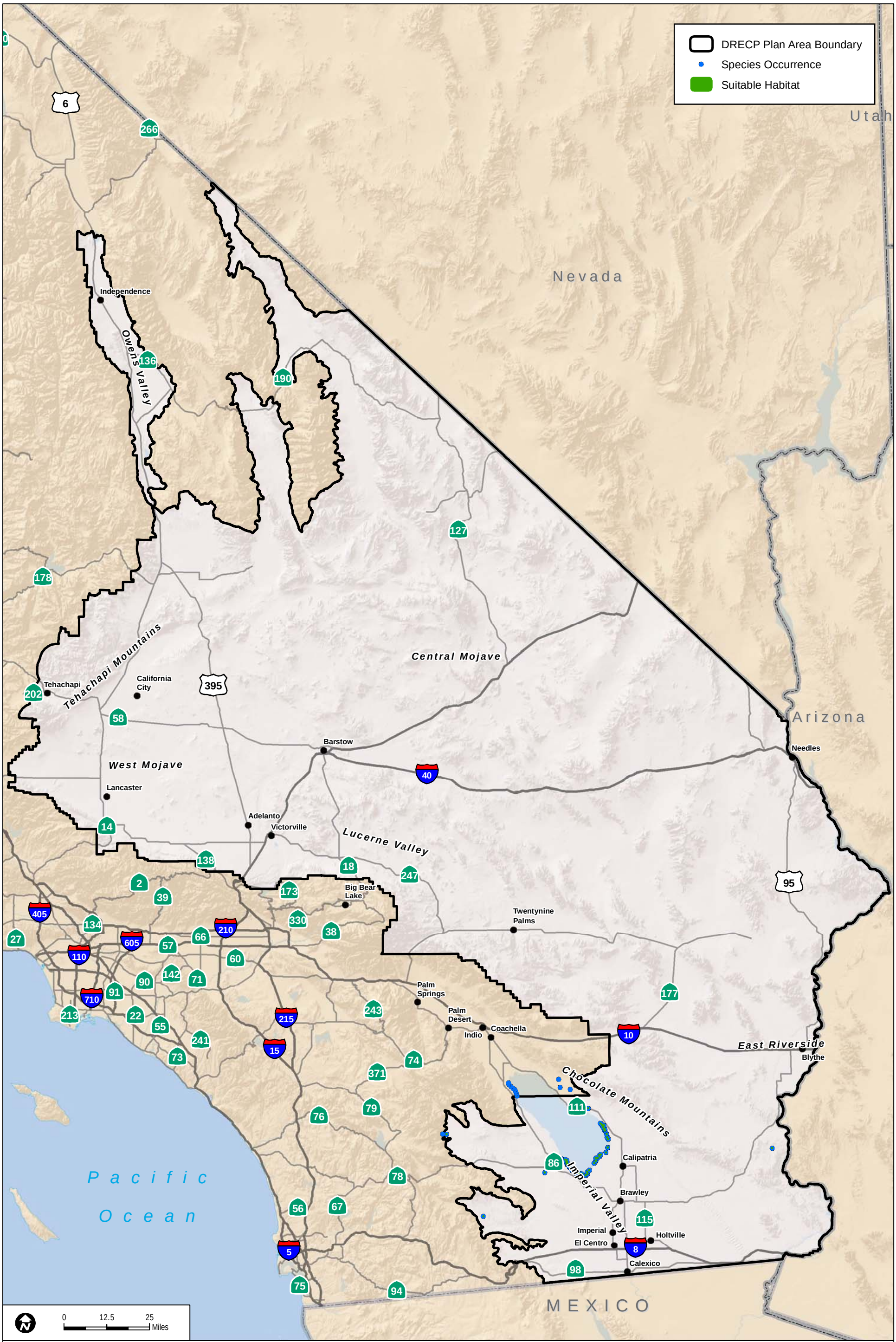
Draft Species Habitat Model Results for Western Yellow-billed Cuckoo



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-B14

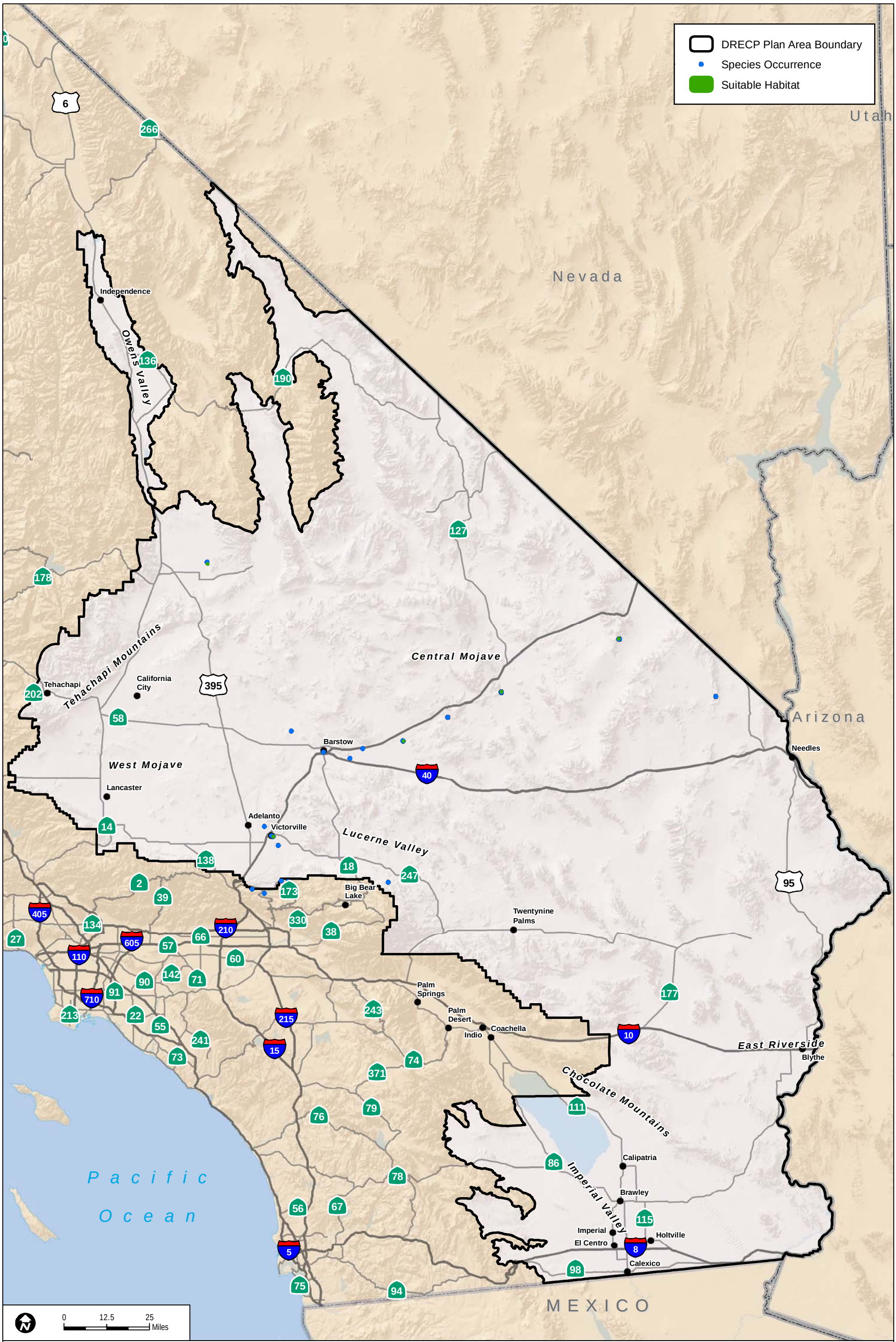
Draft Species Habitat Model Results for Yuma Clapper Rail



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-F01

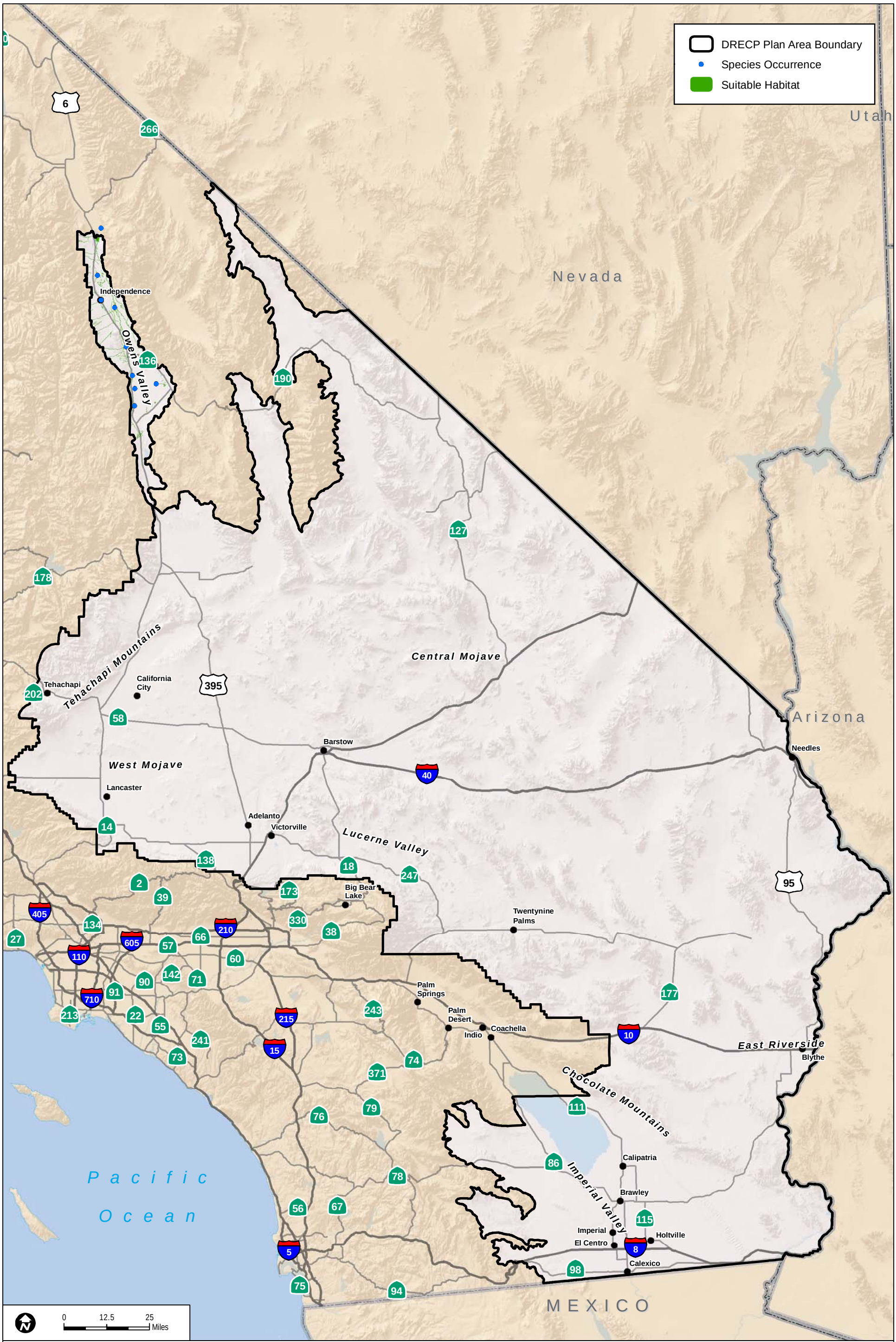
Draft Species Habitat Model Results for Desert Pupfish



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-F02

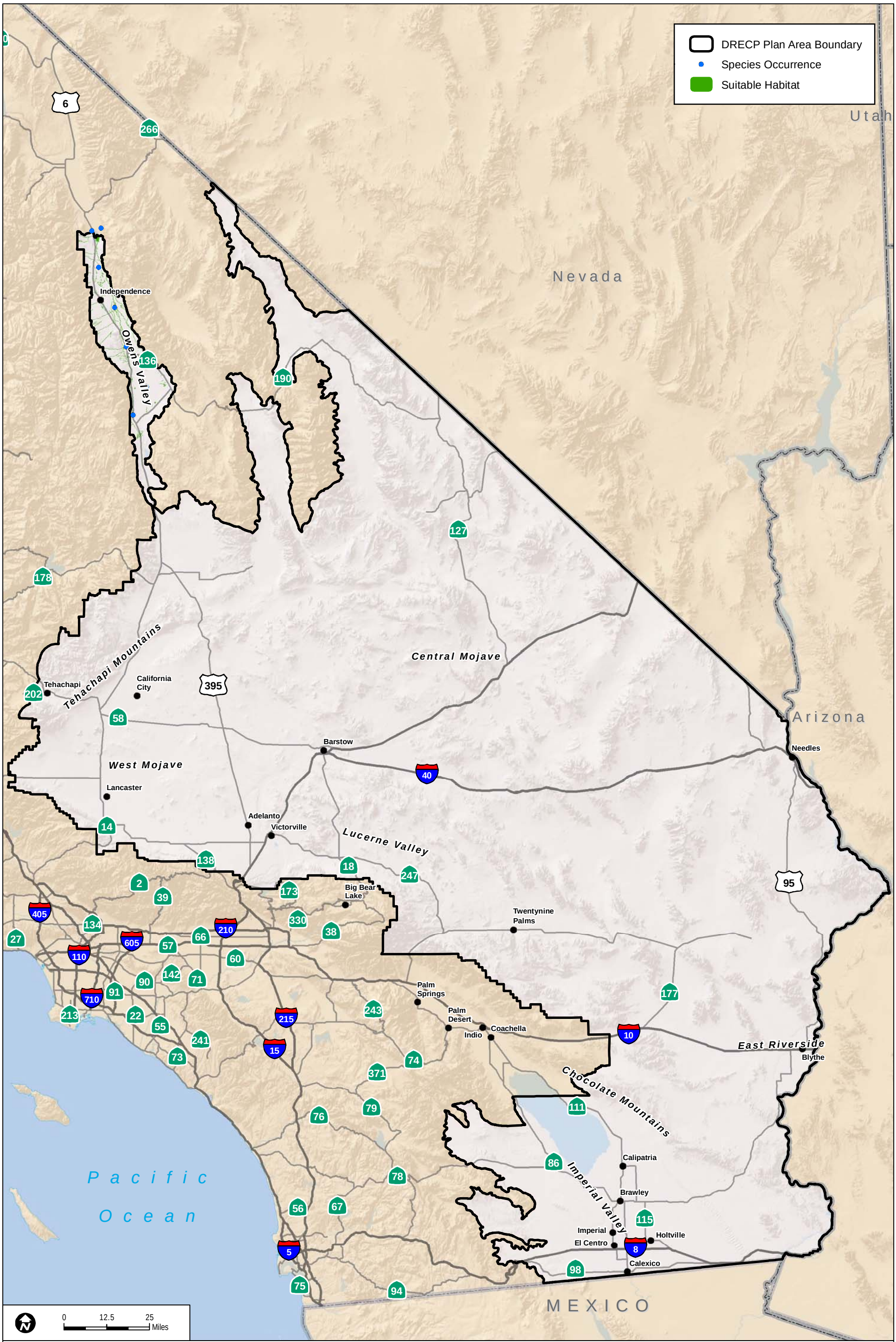
Draft Species Habitat Model Results for Mohave Tui Chub



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-F03

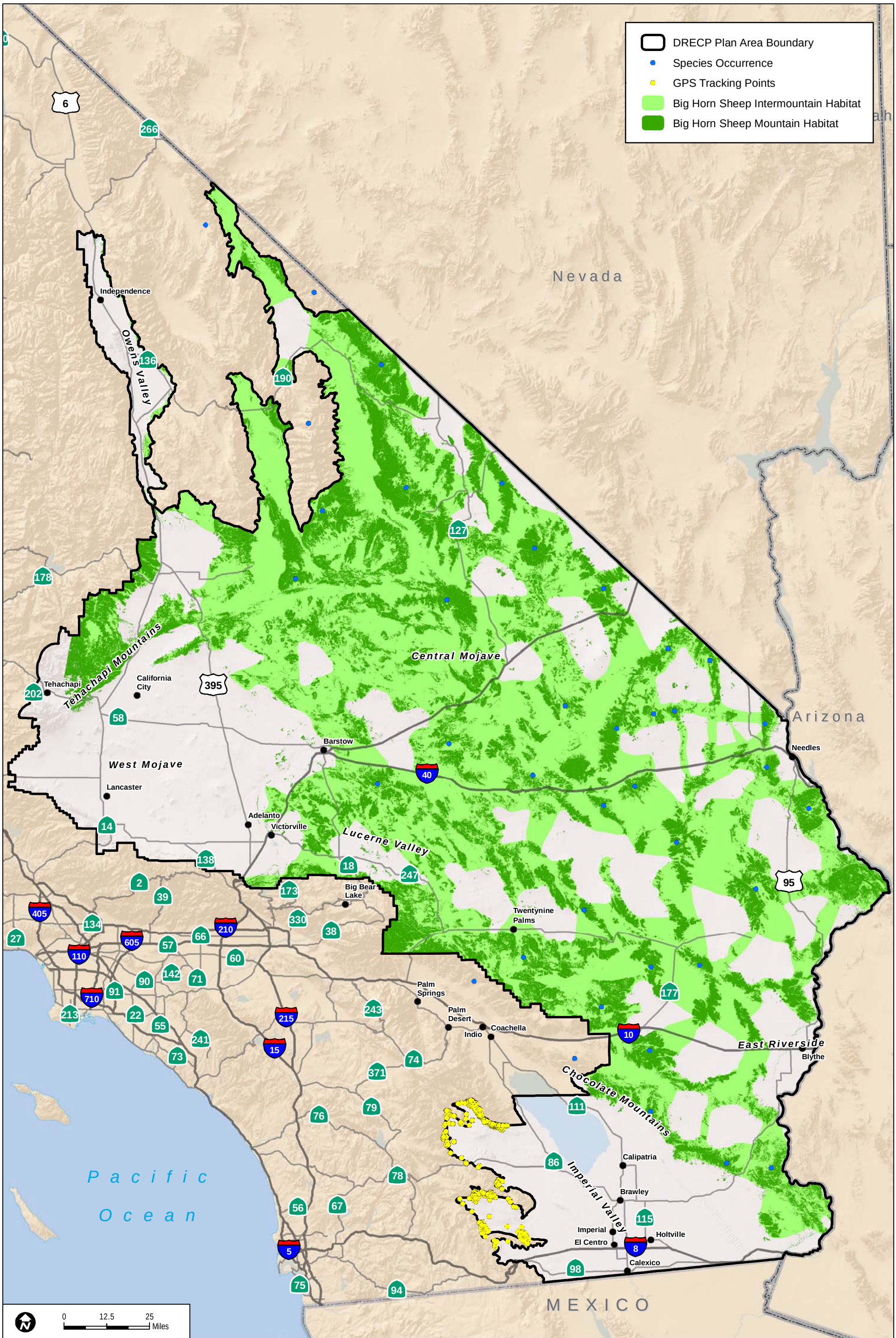
Draft Species Habitat Model Results for Owen's Pupfish



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-F04

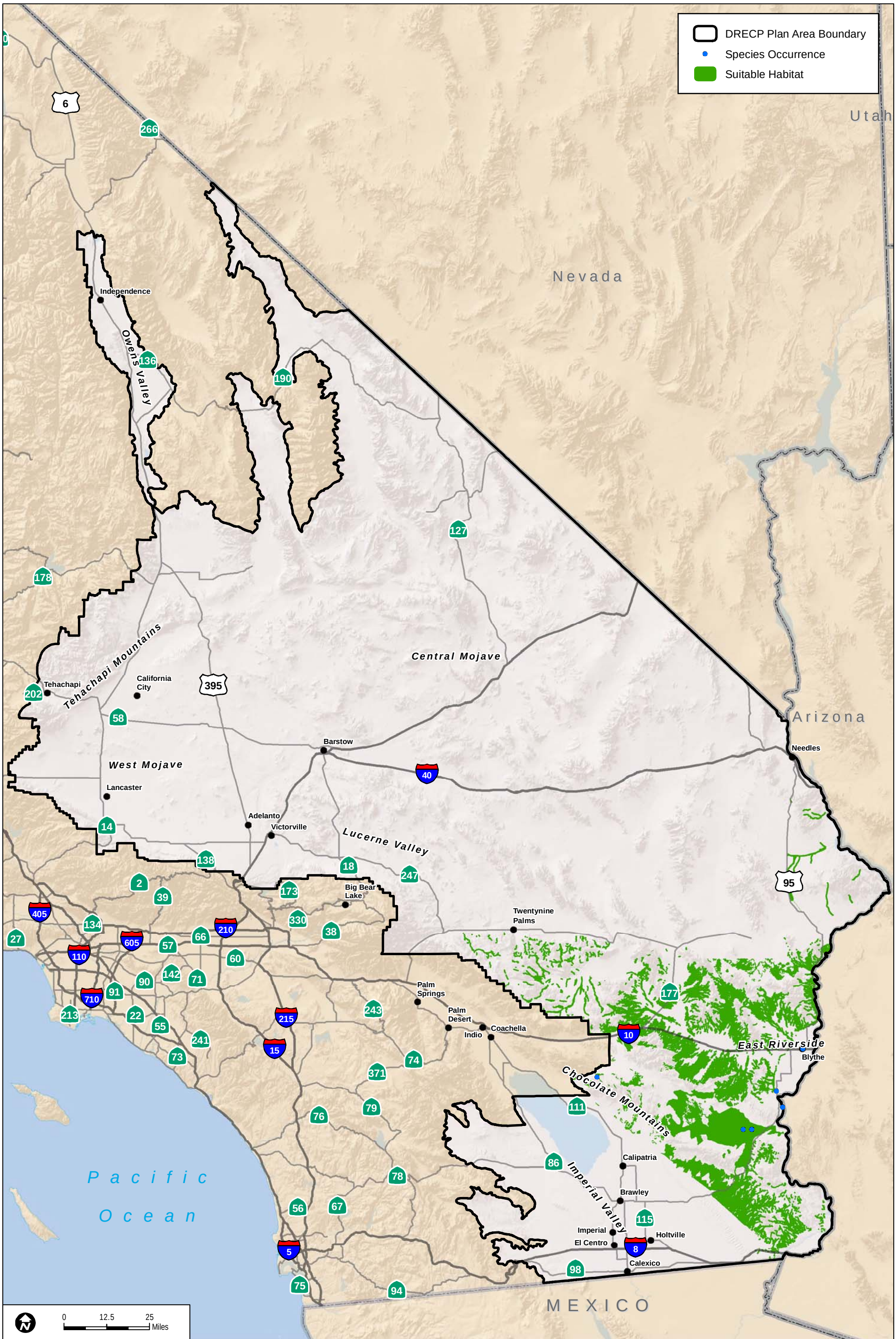
Draft Species Habitat Model Results for Owen's Tui Chub



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); USFWS (2013); CDFW (2013)

FIGURE SM-M01

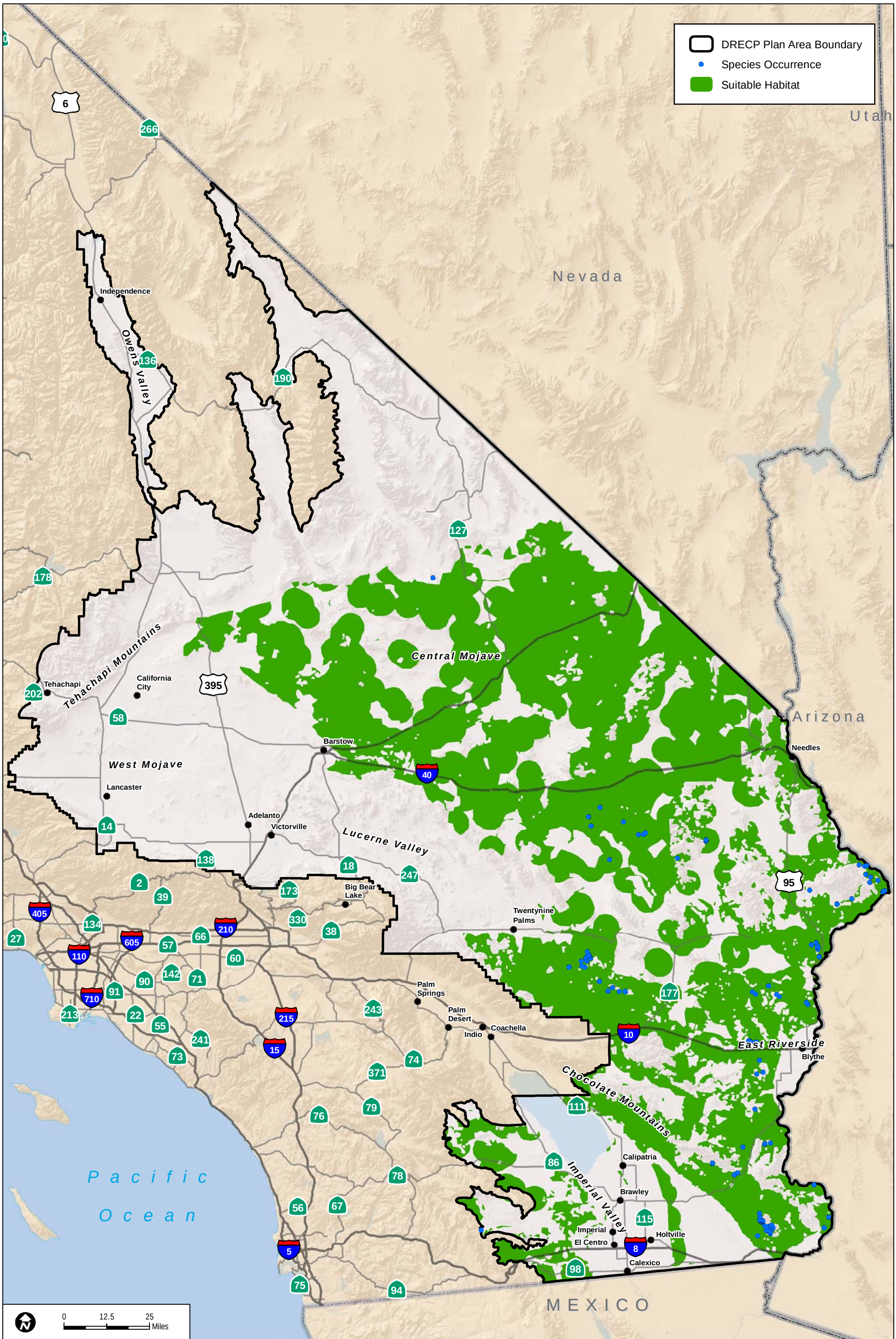
Draft Species Habitat Model Results for Bighorn Sheep Species



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-M02

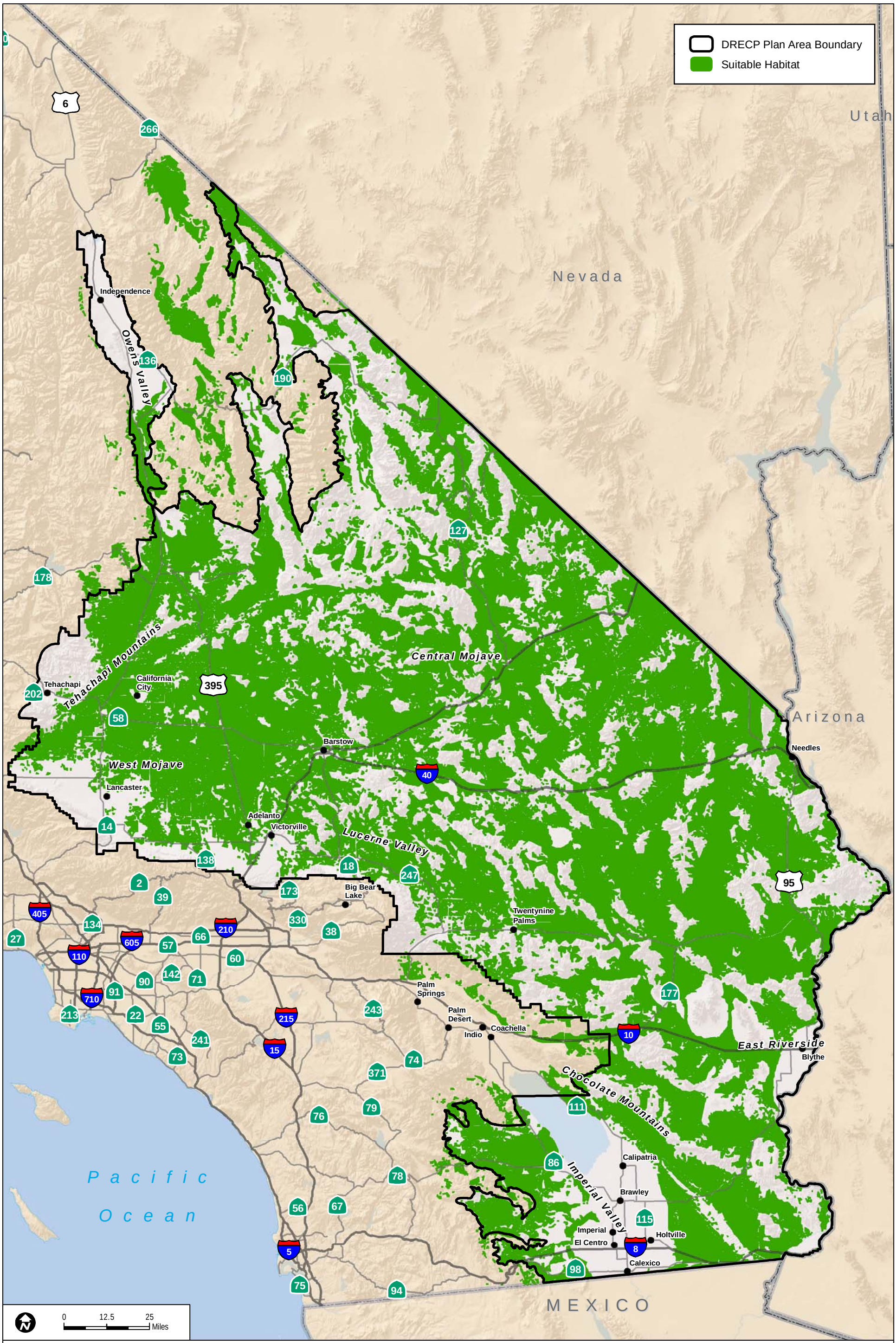
Draft Species Habitat Model Results for Burro Deer



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-M03

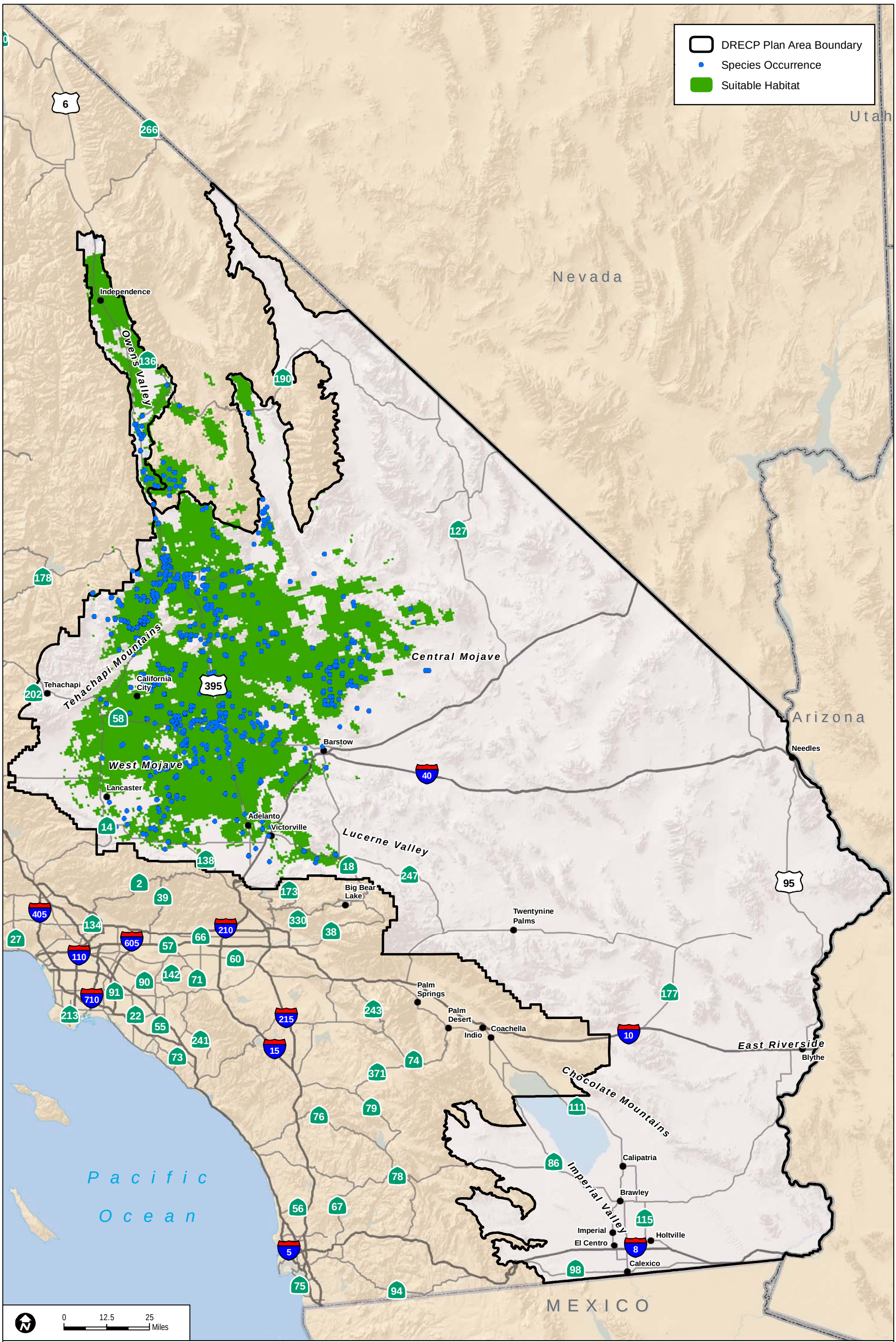
Draft Species Habitat Model Results for Leaf-nosed Bat



Sources: ESRI (2014); CBI (2013)

FIGURE SM-M04

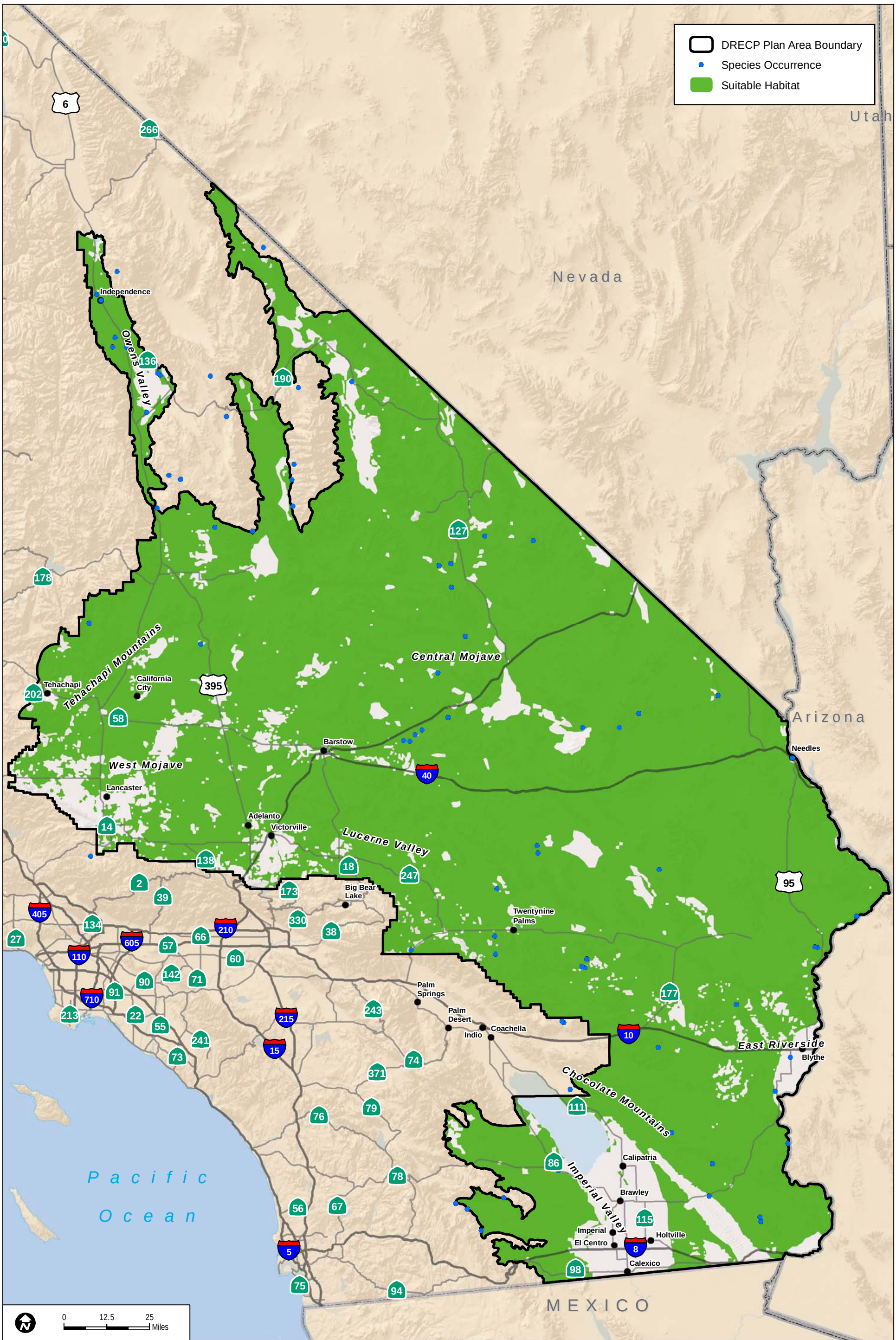
Draft Species Habitat Model Results for Desert Kit Fox



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-M05

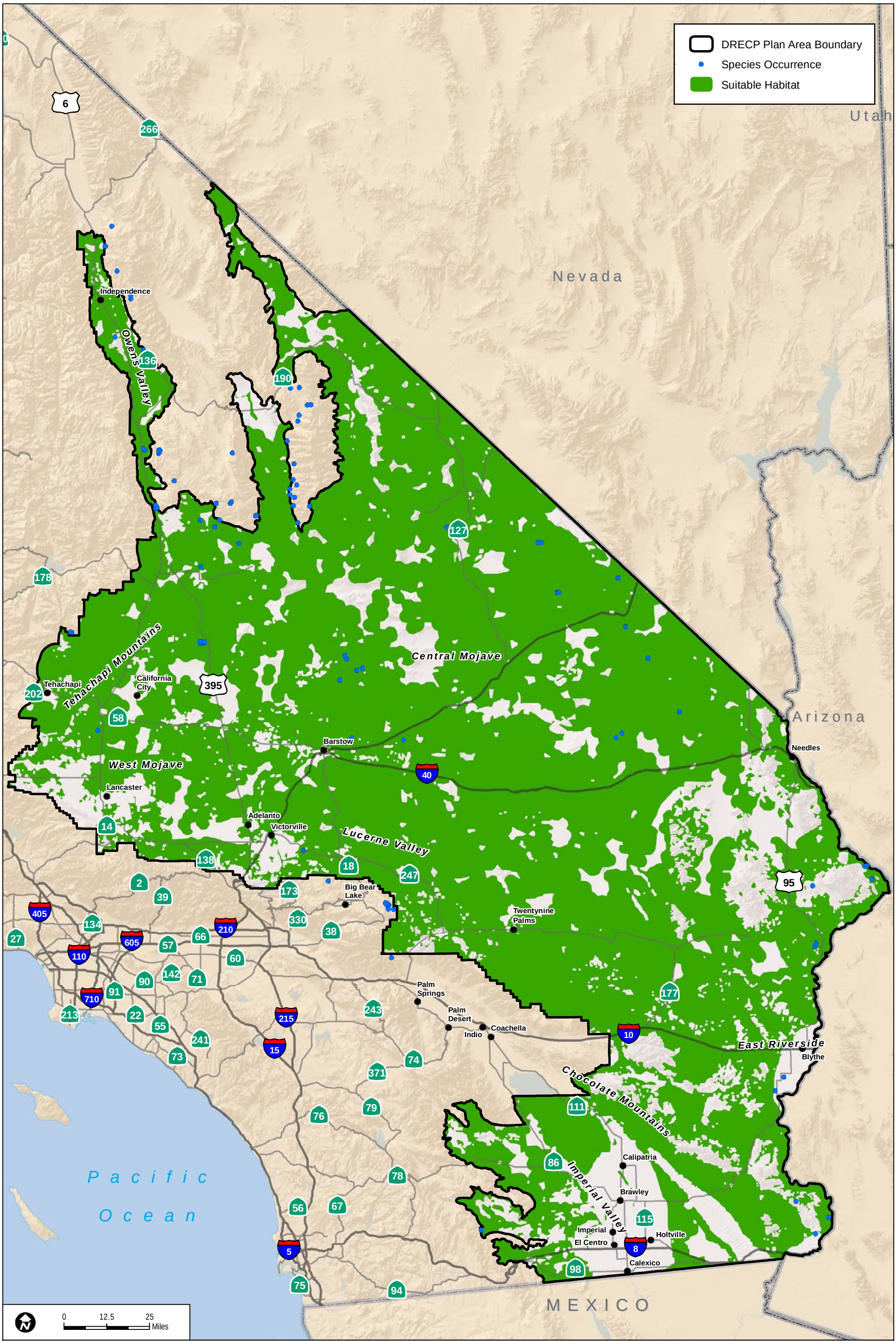
Draft Species Habitat Model Results for Mohave Ground Squirrel



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-M06

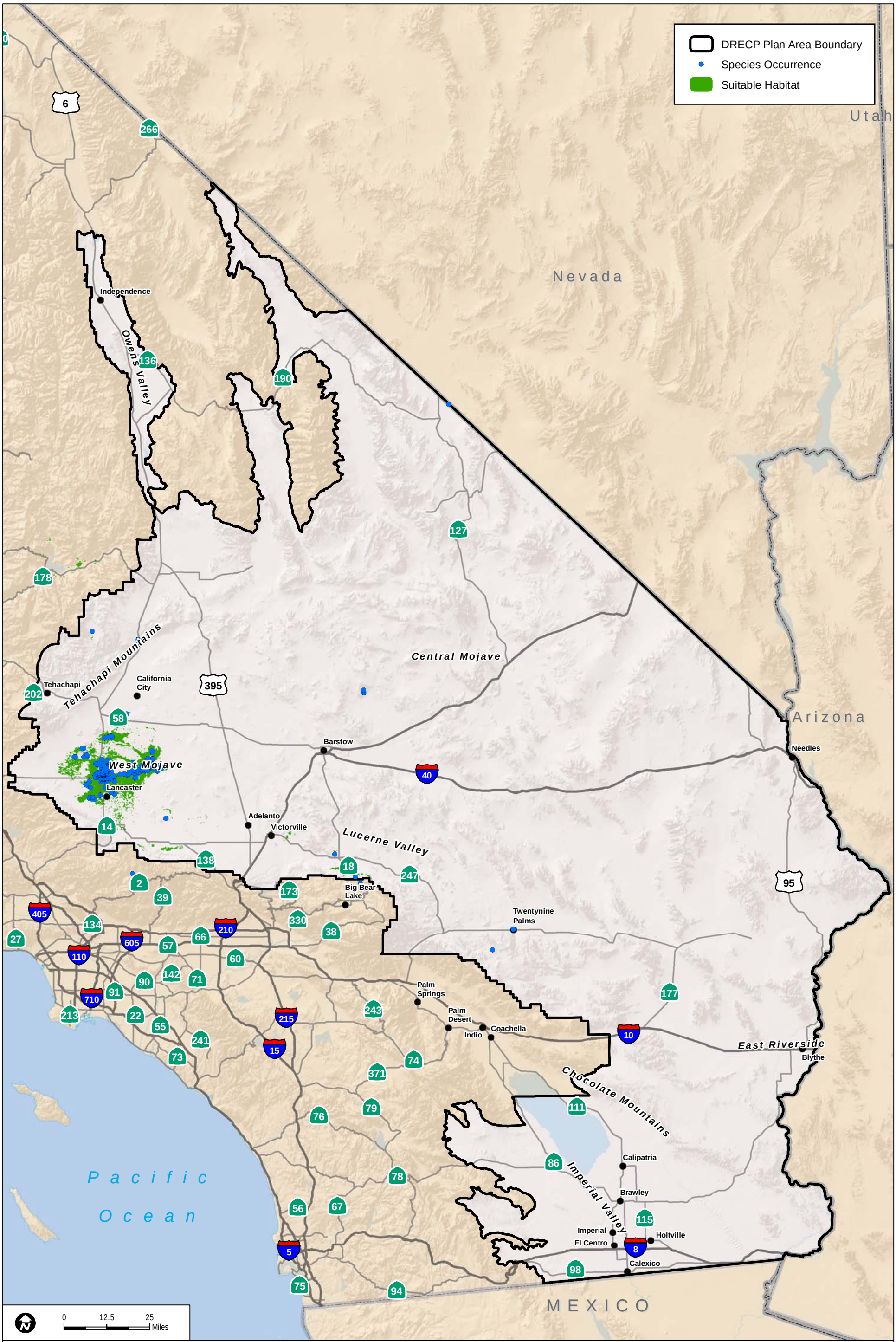
Draft Species Habitat Model Results for Pallid Bat



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-M07

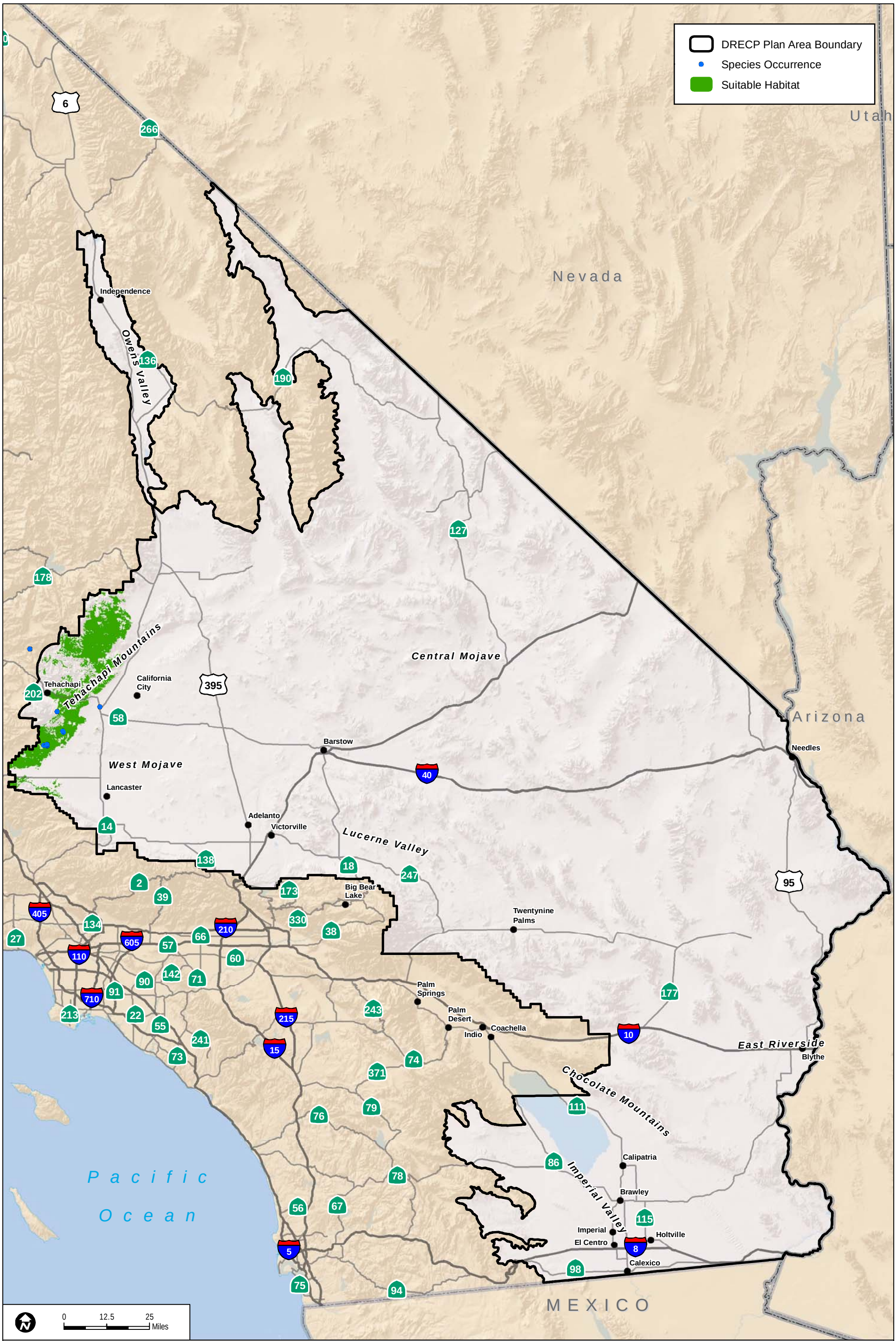
Draft Species Habitat Model Results for Townsends Big-eared Bat



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P01

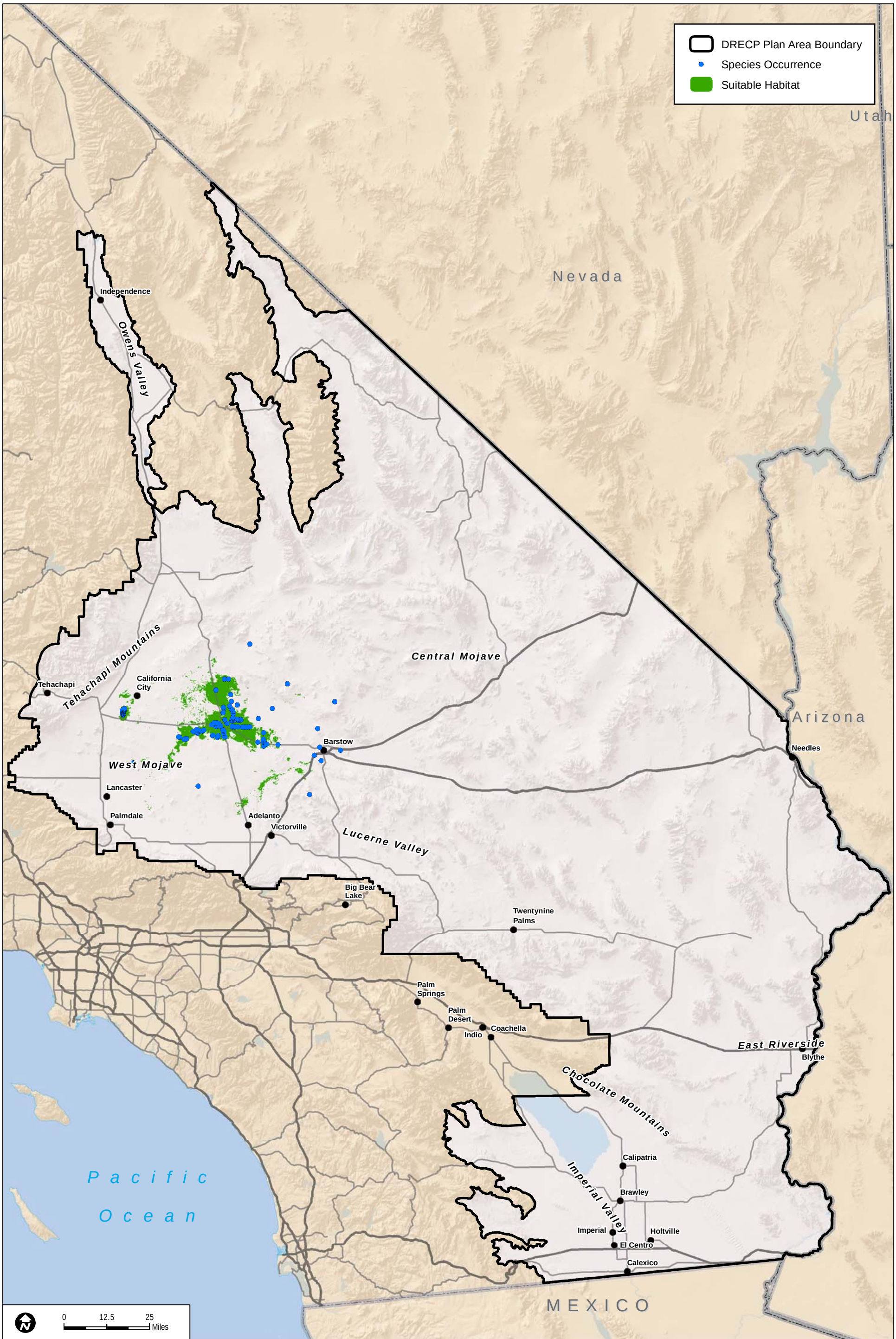
Draft Species Habitat Model Results for Alkali Mariposa Lily



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P02

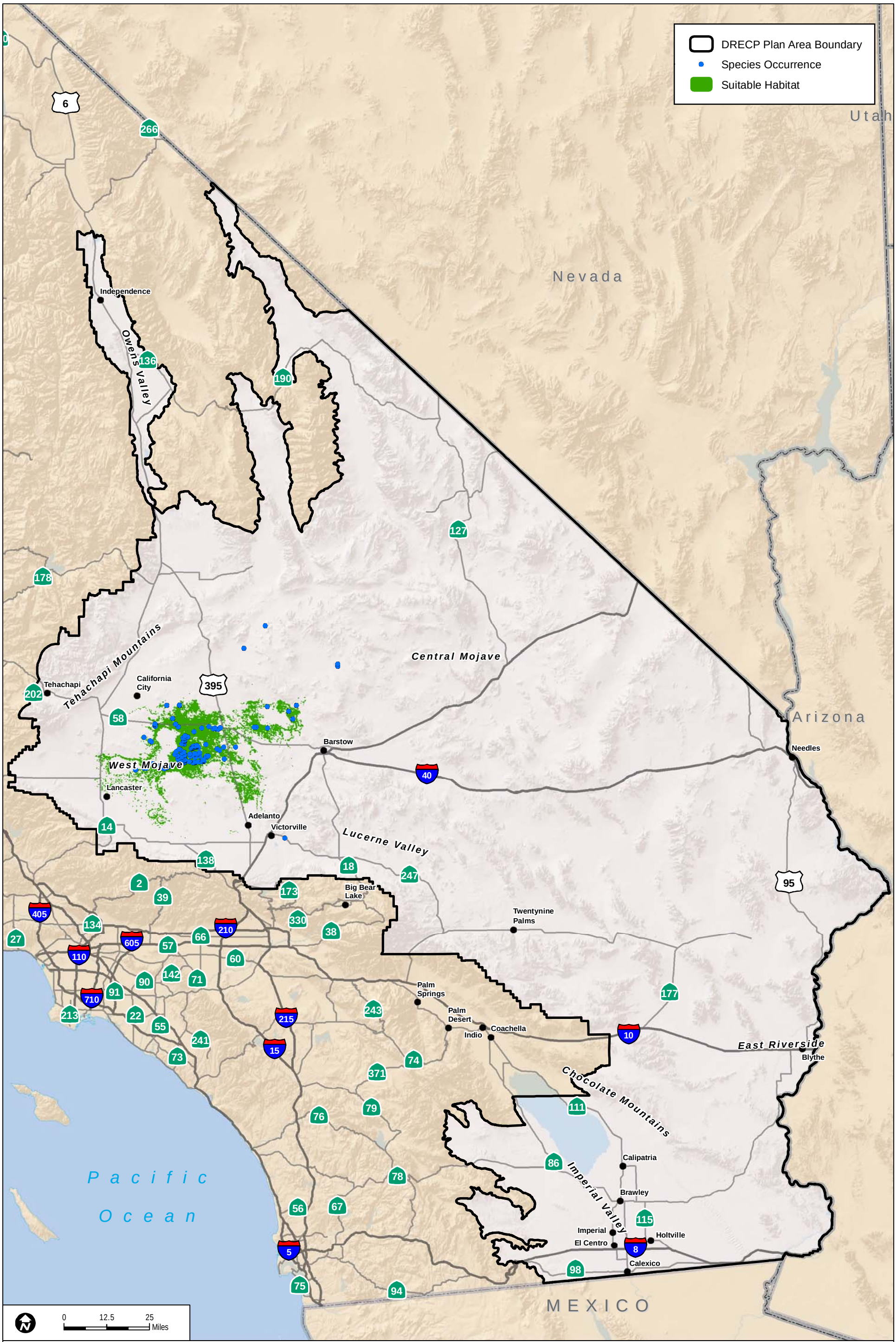
Draft Species Habitat Model Results for Bakersfield Cactus



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P03

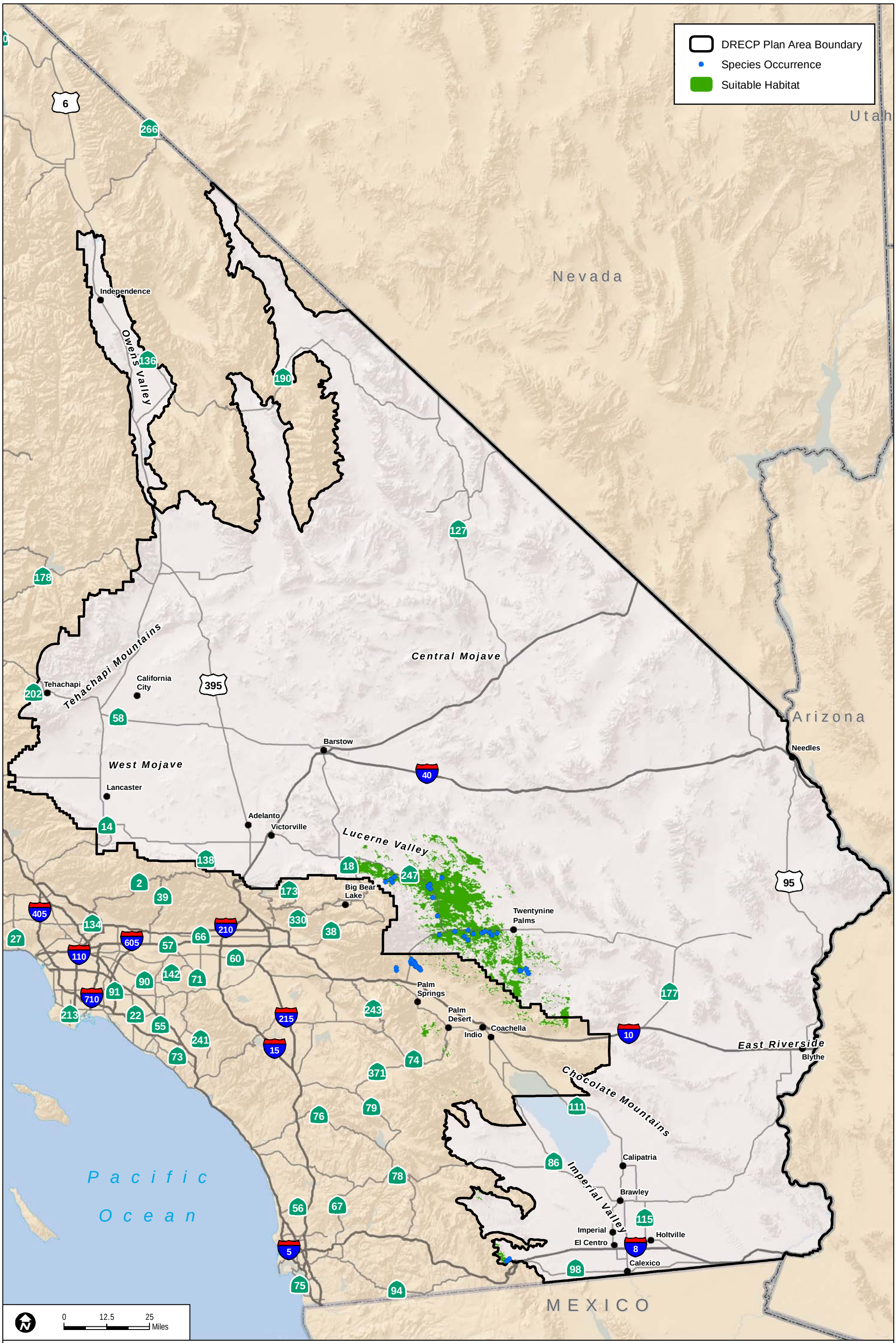
Draft Species Habitat Model Results for Barstow Woolly Sunflower



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P04

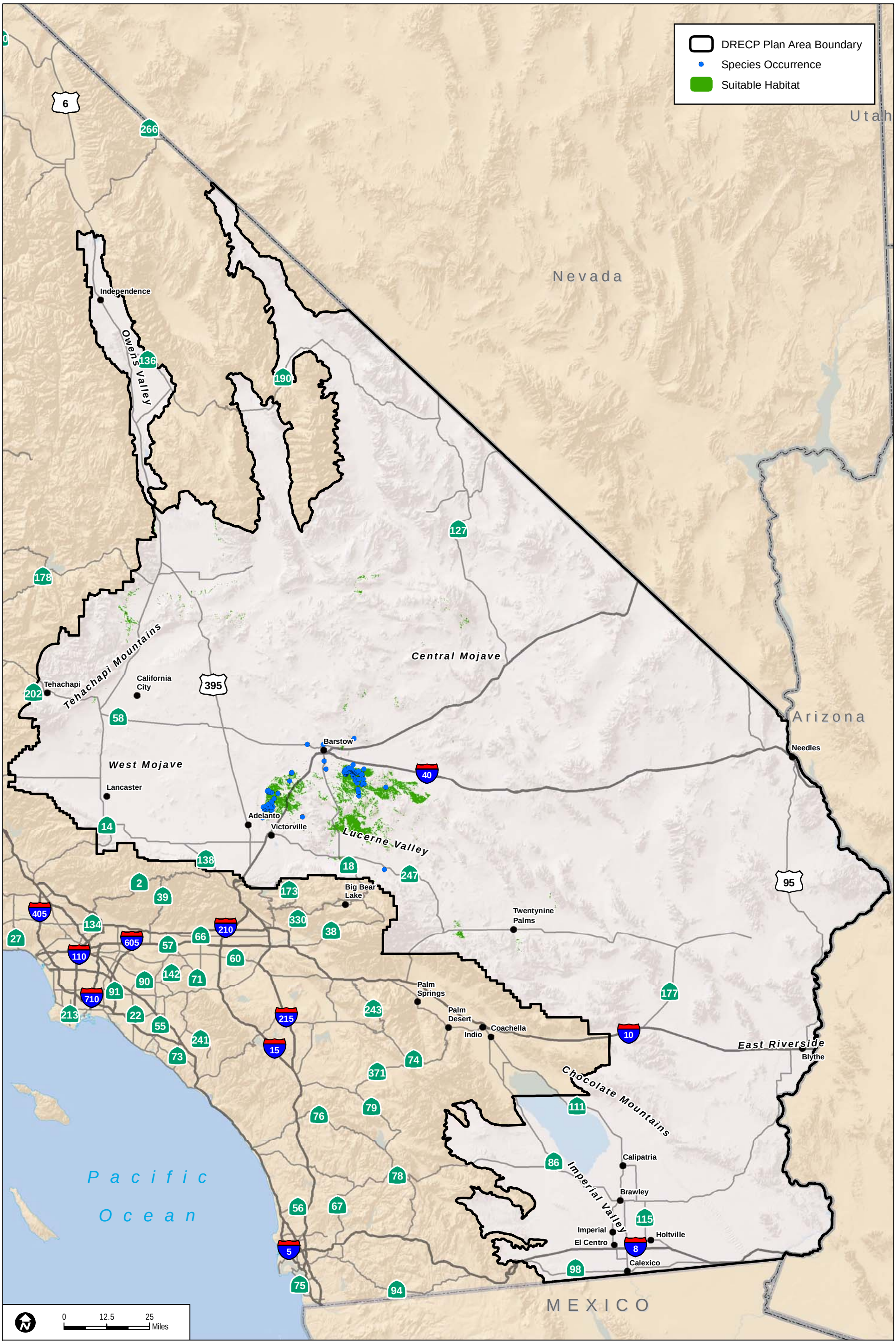
Draft Species Habitat Model Results for Desert Cymopterus



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P05

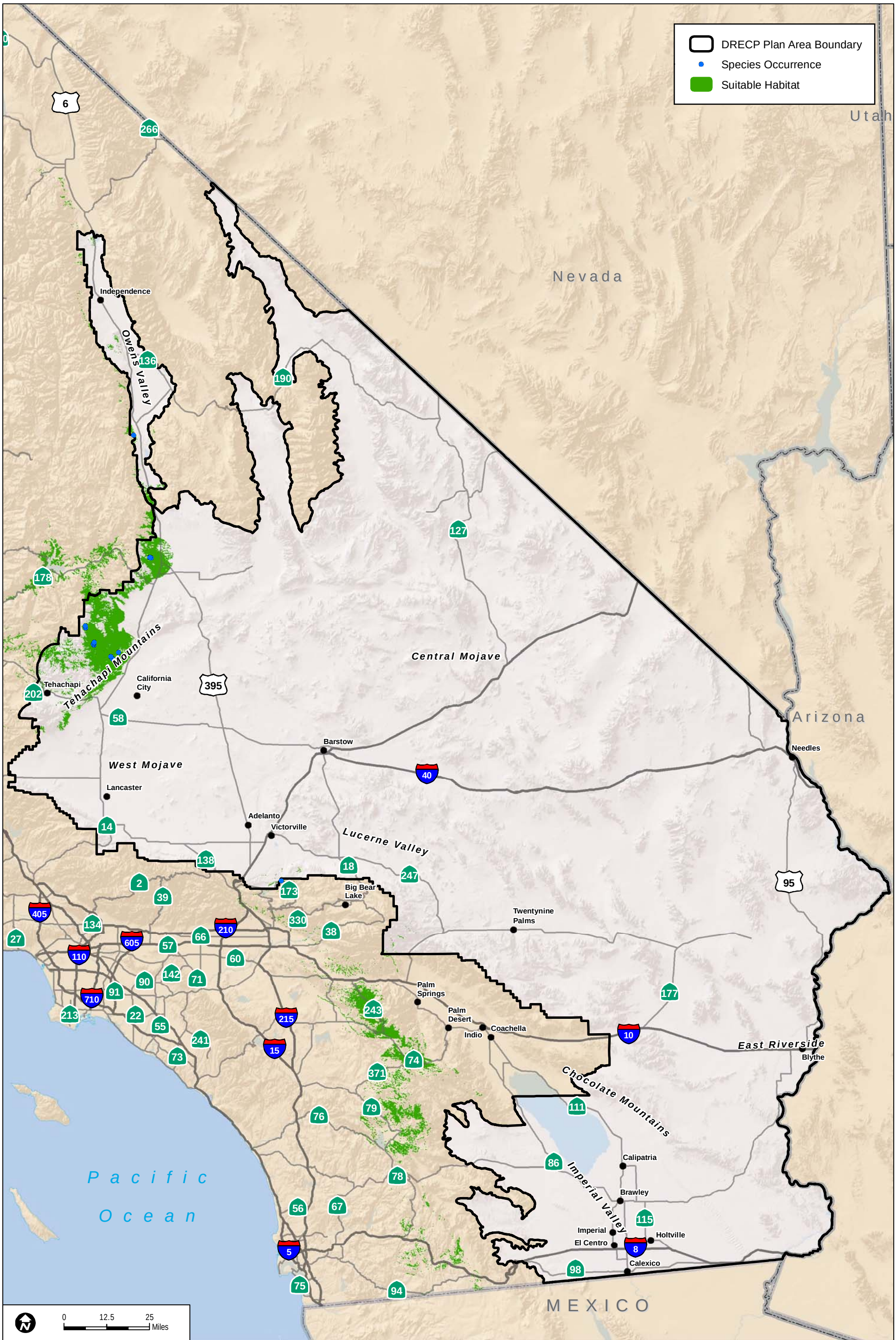
Draft Species Habitat Model Results for Little San Bernardino Mountains Linanthus



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P06

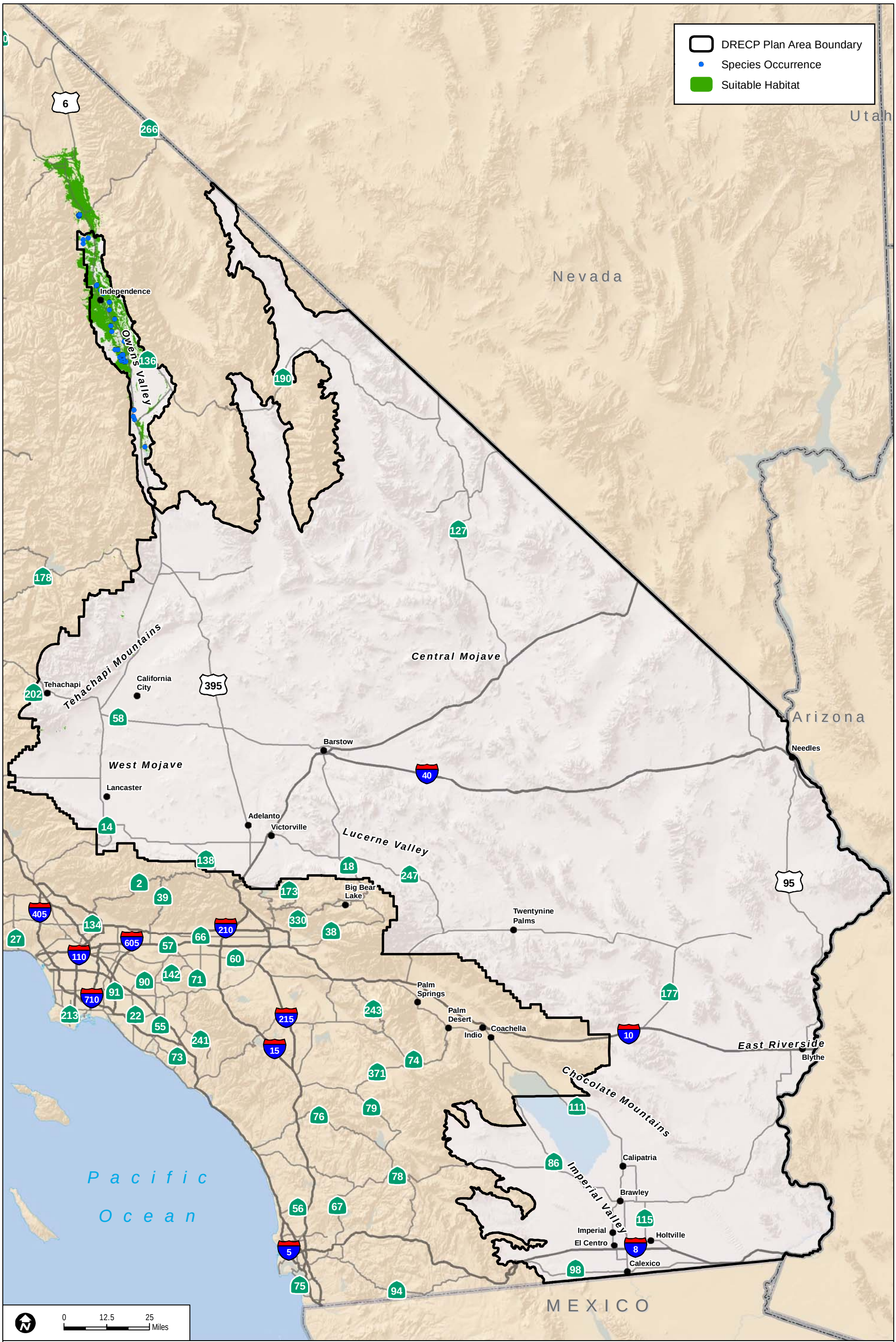
Draft Species Habitat Model Results for Mojave Monkeyflower



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P07

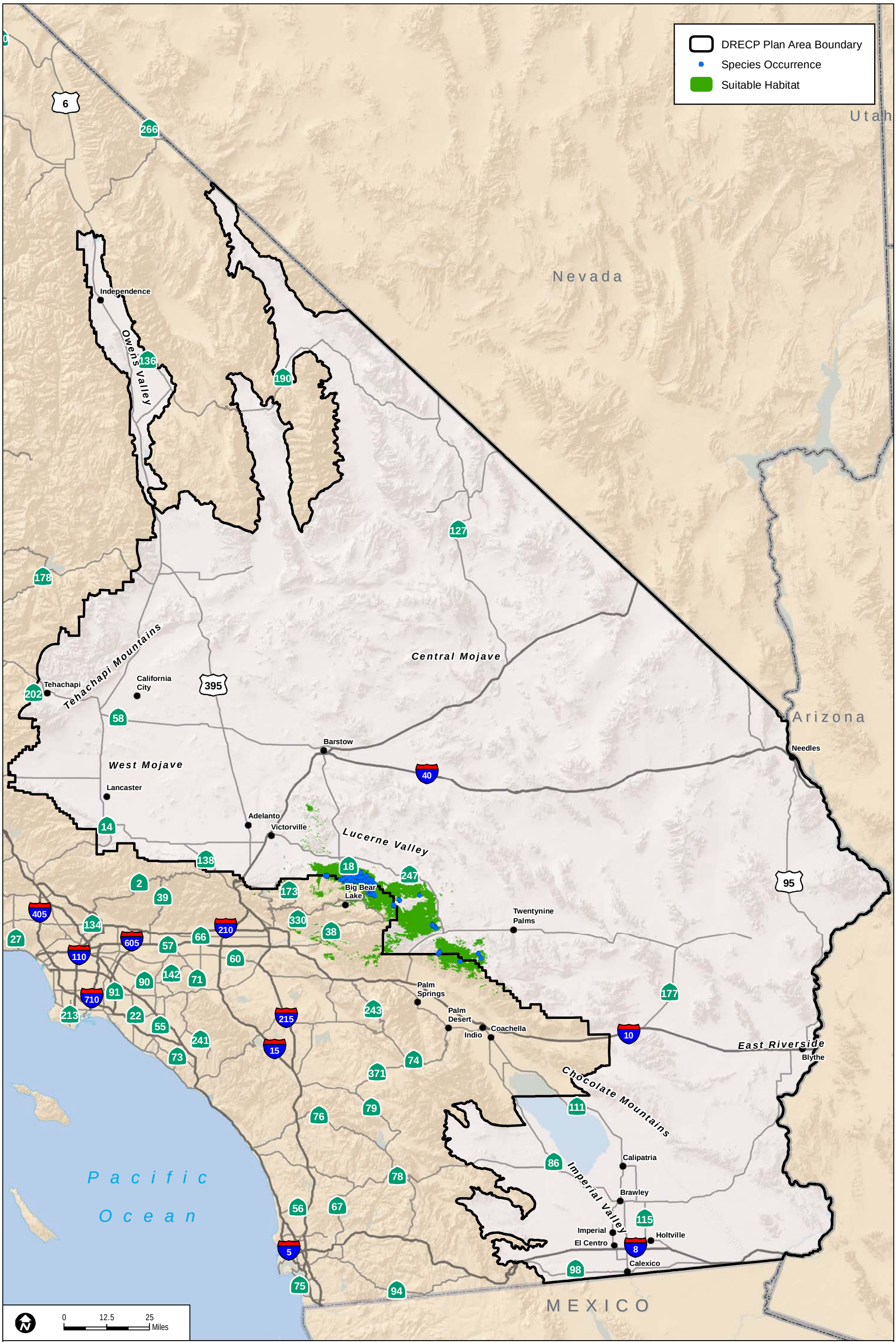
Draft Species Habitat Model Results for Mojave Tarplant



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P08

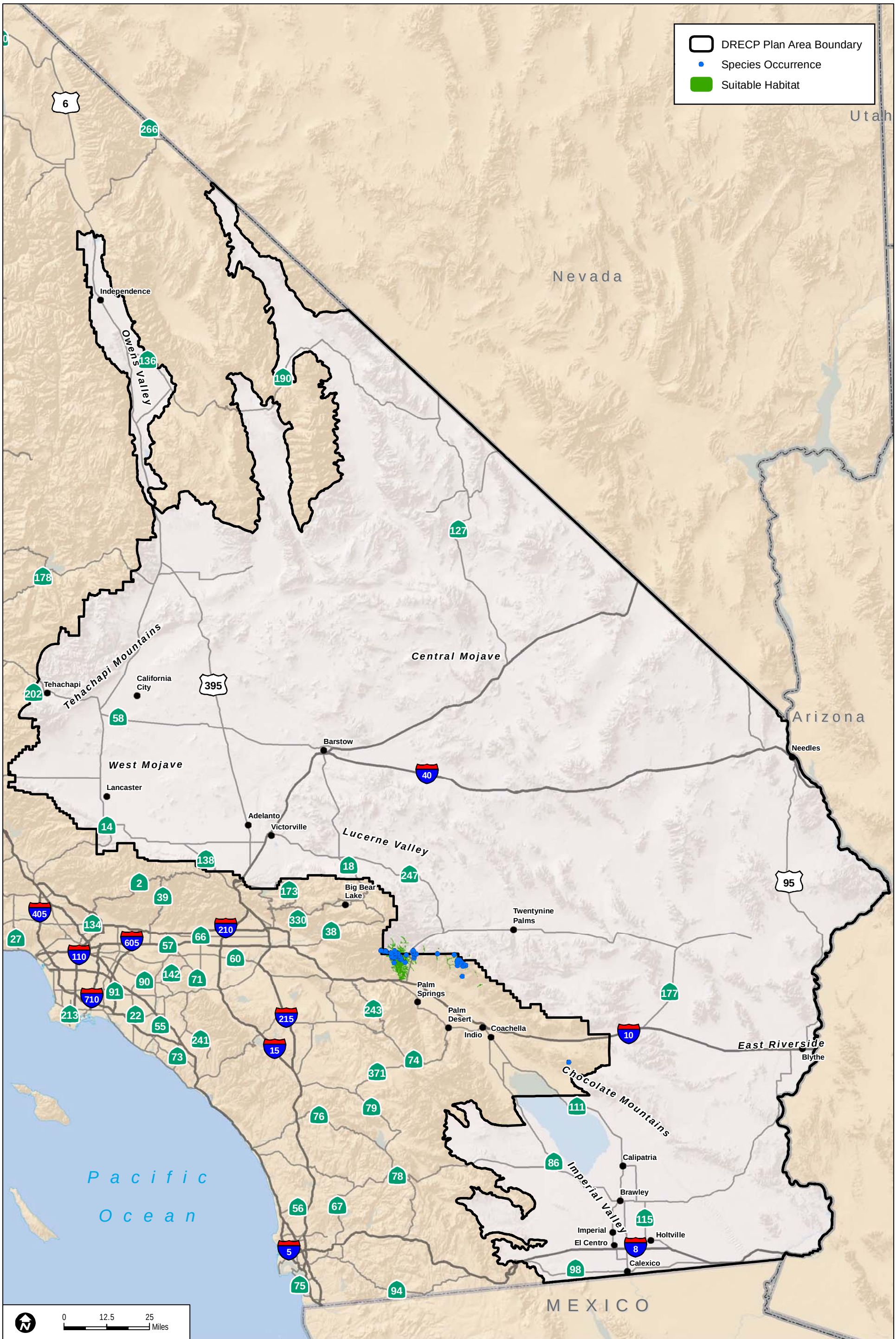
Draft Species Habitat Model Results for Owen's Valley Checkerbloom



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P09

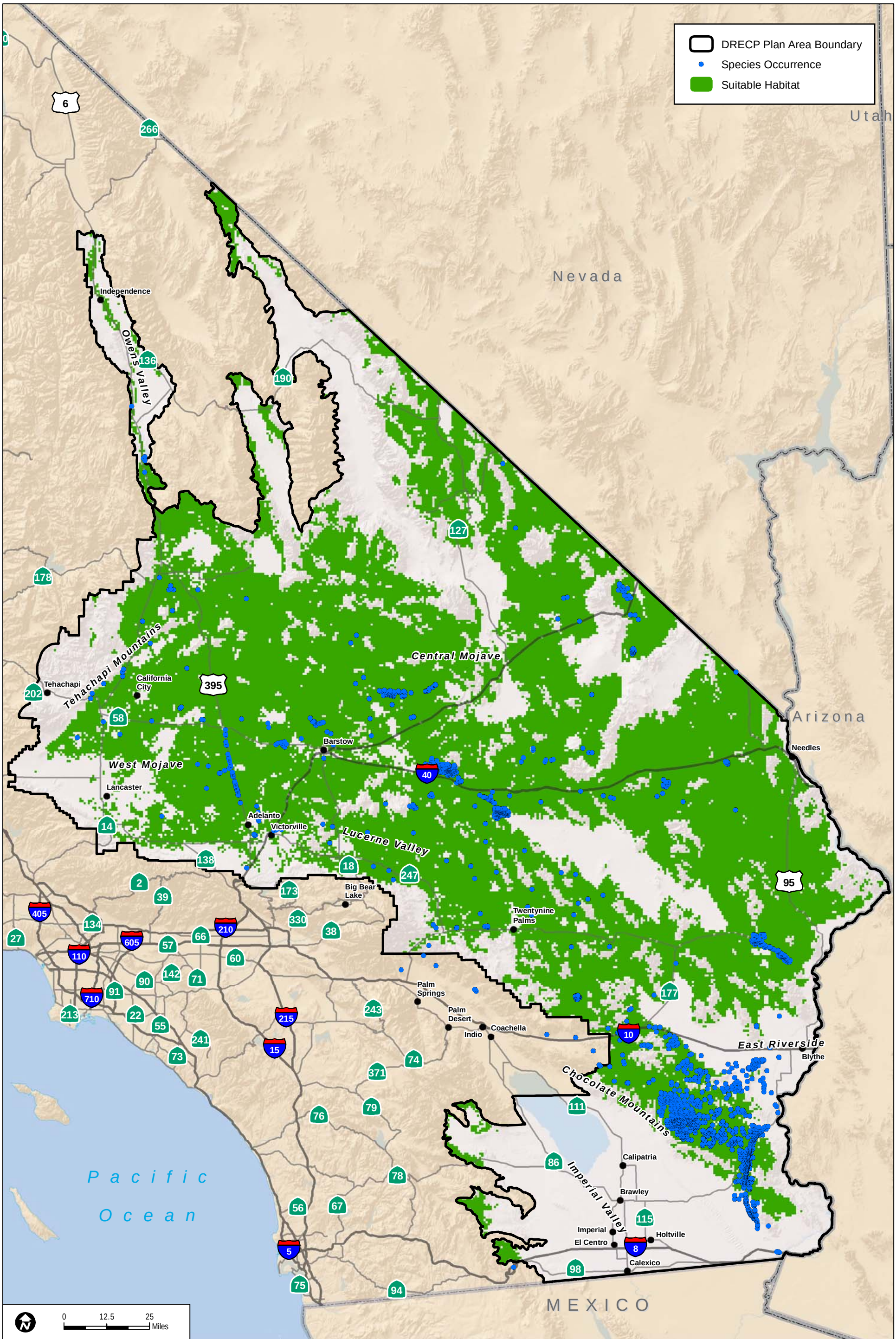
Draft Species Habitat Model Results for Parish's Daisy



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-P10

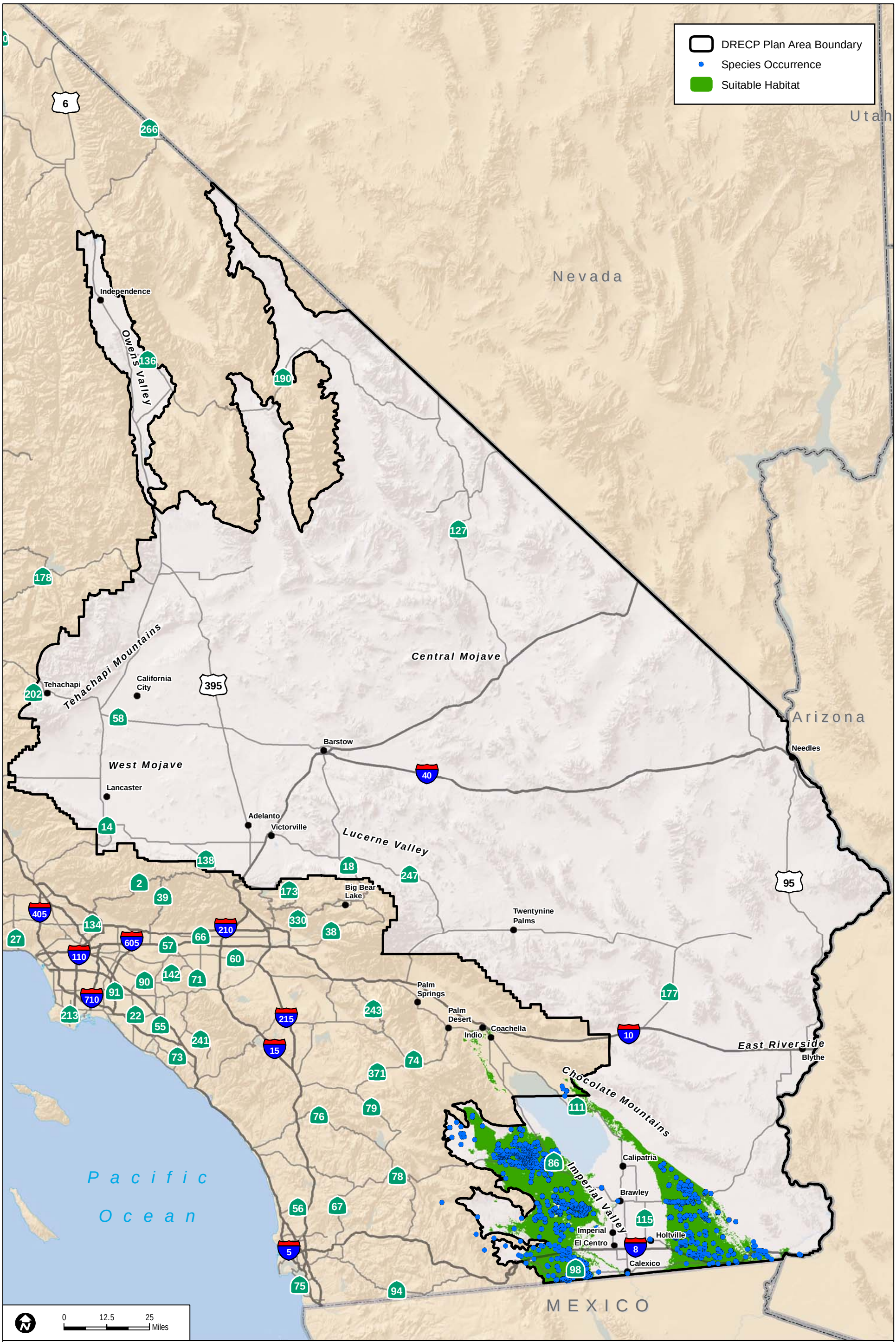
Draft Species Habitat Model Results for Triple-ribbed Milk-vetch



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-R01

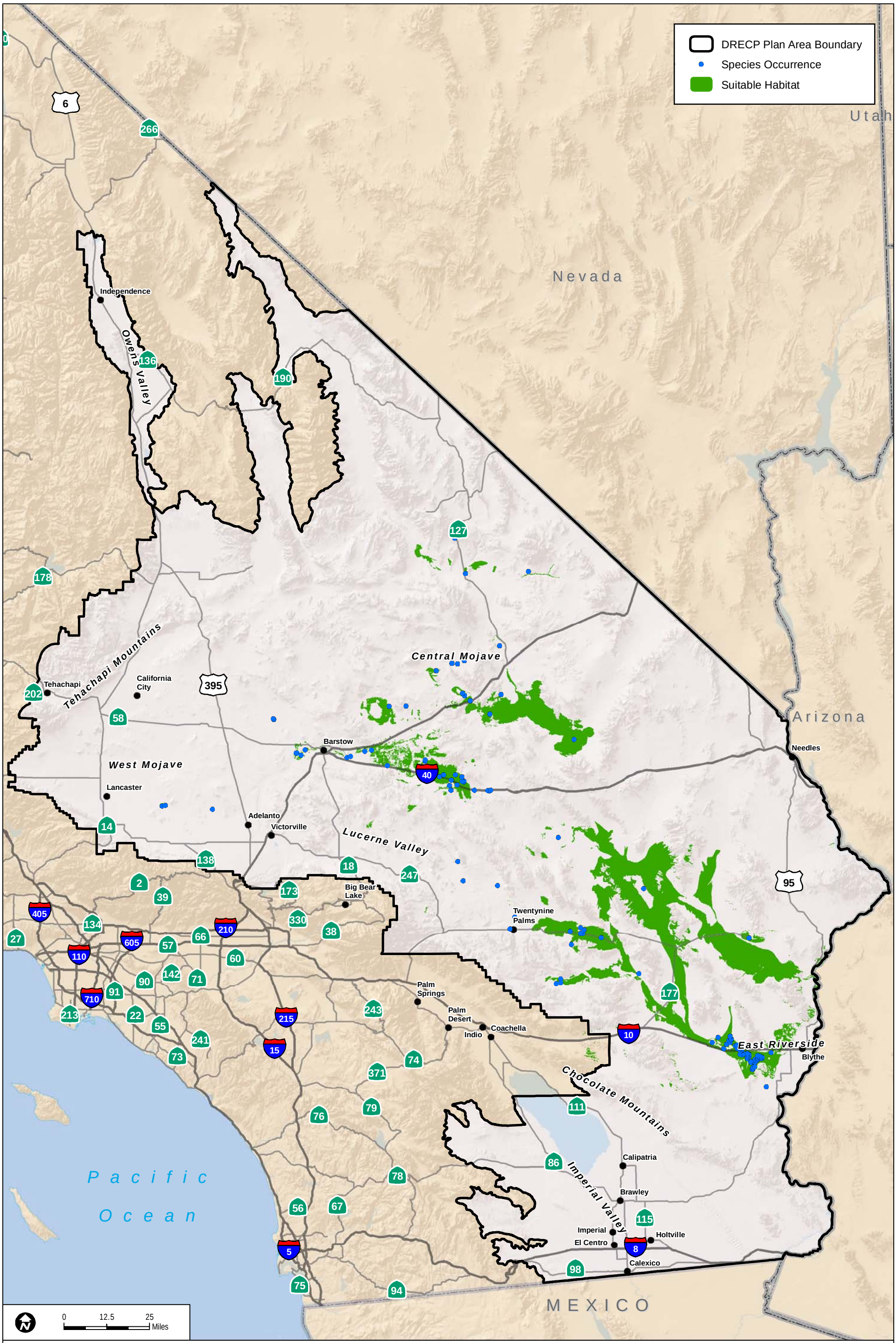
Draft Species Habitat Model Results for Desert Tortoise



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-R02

Draft Species Habitat Model Results for Flat-tailed Horned Lizard



Sources: ESRI (2014); DRECP Species Occurrence Database (2013); CBI (2013)

FIGURE SM-R03

Draft Species Habitat Model Results for Mojave Fringe-toed Lizard