

APPENDIX E-1:
EXISTING BIOLOGICAL CONDITIONS REPORT

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**EXISTING BIOLOGICAL
CONDITIONS REPORT**

for the

**Upper Santa Ana River
Habitat Conservation Plan**

Prepared for:

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Water Conservation District

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EXECUTIVE SUMMARY

The Upper Santa Ana River Wash Land Management Plan and Habitat Conservation Plan (USAR HCP) has been in development by the stakeholders, which include the San Bernardino Valley Water Conservation District (SBVWCD), Cemex Construction Materials, L.P. (Cemex), Robertson's Ready Mix (Robertson's), the City of Highland, the City of Redlands, the San Bernardino County Department of Transportation, Flood Control District (SBCO FCD), and in cooperation with the Bureau of Land Management (BLM), U.S. Fish and Wildlife Service (Service) and the California Department of Fish and Game (Department) for several years. The SBVWCD is the Lead Agency for the stakeholder group. The Upper Santa Ana Wash Plan Area consists of approximately 4,491 acres and includes lands within the jurisdiction of the County of San Bernardino (County), the Cities of Highland and Redlands, and the BLM. The Plan Area is roughly bounded by Greenspot Road to the north and east, the south bank of the Santa Ana River on the south, and Alabama Street to the west.

The SBVWCD has applied for a permit from the U.S. Fish and Wildlife Service (Service) pursuant to section 10(a)(1)(B) of the Endangered Species Act of 1973 (Act) as amended (16 U.S.C. 1531 *et seq.*) to incidentally take 15 species of which the San Bernardino kangaroo rat (*Dipodomys merriami*, "SBKR"), Santa Ana Woolly-star (*Eriastrum densifolium* ssp. *sanctorum*, "woollystar"), slender-horned spineflower (*Dodecahema leptoceras*, "spineflower") are state and federally listed as endangered and the coastal California gnatcatcher (*Polioptila californica californica*, "gnatcatcher") is federally listed as threatened. The Applicant is also requesting incidental take authorization for 11 species that are either considered Sensitive Species by the Bureau of Land Management (BLM), California Special Concern Species and/or are otherwise identified as sensitive species (see Section 4.0, Table 1 below).

This report describes the existing biological condition of the wash plan area as they pertain to the proposed covered species. Information regarding biological resources is taken from past biological survey reports provided by the various participating agencies and some minor, recent supplemental field work conducted by Dudek. Resources described in this report include physical characteristics of the site (including soils, land use, topography, hydrology), vegetation communities, and species descriptions for each of the proposed covered species.

1.0 INTRODUCTION

This report provides a description of biological resources within the proposed Upper Santa Ana River Habitat Conservation Plan (USAR HCP) Wash Plan Areas. The USAR HCP proposed incidental take of four federally-listed plant or animal species (San Bernardino kangaroo rat [*Dipodomys merriami*, "SBKR"], Santa Ana woolly-star [*Eriastrum densifolium* ssp *sanctorum*, "woollystar"], slender-horned spineflower [*Dodecahema leptoceras*, "spineflower"] and the coastal California gnatcatcher [*Poliptila californica californica*, "gnatcatcher"]), and 11 other sensitive plant and animal species. This report focuses on resources that pertain to the life history of these species. All species for which incidental take permits would be issued are presented in *Table 1*.

TABLE 1. COVERED SPECIES

COMMON NAME	SCIENTIFIC NAME	STATUS FEDERAL/STATE
AMPHIBIANS & REPTILES		
Coast or San Diego horned lizard	<i>Phrynosoma coronatum blainvillei</i>	CSC
Western spadefoot	<i>Spea hammondi</i>	BLM/CSC
BIRDS		
Cactus wren	<i>Campylorhynchus brunneicapillus</i>	BCC/CSC
Coastal California gnatcatcher	<i>Poliptila californica californica</i>	FT/ CSC
Loggerhead shrike	<i>Lanius ludovicianus</i> (nesting)	BCC, MNBMC/CSC
Southern California rufous-crowned sparrow	<i>Aimophila ruficeps canescens</i>	None/ CSC
Western burrowing owl	<i>Athenecunicularia</i>	BCC, BLM, MNBMC/CSC
MAMMALS		
Los Angeles pocket mouse	<i>Perognathus longimembris brevinasus</i>	CSC
San Bernardino kangaroo rat	<i>Dipodomys merriami parvus</i>	FE/CSC
FISH		
Speckled dace	<i>Rhinichthys osculus</i> ssp.3	CSC
PLANTS		
Parry's spineflower	<i>Chorizanthe parryi</i>	CNPS List 1B
Plummer=s mariposa lily	<i>Calochortus plummerae</i>	CNPS List 1B
Robinson=s pepper-grass	<i>Lepidium virginicum</i> var. <i>robinsonii</i>	CNPS List 1B
Santa Ana River woollystar	<i>Eriastrum densifolium</i> ssp. <i>sanctorum</i>	FE/SE/CNPS List 1B
Slender-horned spineflower	<i>Dodecahema leptoceras</i>	FE/SE/CNPS List 1B

Federal Designations:

BCC	Birds of Conservation Concern (U.S. Fish and Wildlife Service)
BLM	Bureau of Land Management Sensitive Species
FC	Candidate for federal listing as Threatened or Endangered
FE	Federally-listed Endangered
FT	Federally-listed Threatened
MNBMC	Fish and Wildlife Service Migratory Nongame Birds of Management Concern

State Designations:

CSC	California Special Concern Species
SE	State-listed as Endangered

Special Designations:

CNPS List 1B	California Native Plant Society Highest priority endangerment list
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1.1 Plan Area

The Wash Planning Area is located within the alluvial fan of the Santa Ana River, extending approximately six miles between Greenspot Road in the City of Highland on the east, to Alabama Street in the City of Redlands on the west. The Wash Planning Area encompasses approximately 4,491 acres and includes lands within the jurisdiction of the County of San Bernardino (County), the Cities of Highland and Redlands, and the BLM (*Figures 1 and 2*). The project area is approximately one mile downstream of the recently-completed Seven Oaks Dam. The project area is within Sections 7, 8, 17, and 18 of Township 1 South, Range 2 West and Sections 9,10,11, 12, 13, 14, 15, and 16 of Township 1 South, Range 1 West on the San Bernardino South and Yucaipa 7.5 minute United States Geologic Service topographic maps.

1.2 Information Reviewed

Information regarding biological resources of this wash plan area was obtained through an extensive literature search utilizing public databases, private consultant reports, and scientific journal articles. The first level of review pertained to understanding the existing biological conditions of the was plan area; the second level of review was focused on understanding species characters and requirements for those species proposed for incidental take under the proposed USAR HCP. Supplemental field surveys were conducted by Dudek but were focused on specific areas and resources and did not include any comprehensive review of the site.

A complete list of references is provided in Section 7.0; summarized here are references related to documenting the existing biological conditions of the site. Public database resources reviewed included Bureau of Land Management South Coast Resources Plan (BLM 1994), California Natural Diversity Database (CNDDDB 2003), USFWS species occurrence data (USFWS 2003), and the area soil survey (Knecht 1971). Previous biological survey reports include a management plan related to woollystar commissioned by U.S. Army Corps of Engineers (ACOE) (Chambers Group 1993), surveys of the Sunwest Material's and Robertson's Ready Mix project areas commission by Lilburn Corporation (Lilburn 1996 and 1997), an Environmental Impact Report for the Metropolitan Water District of Southern California (MWD) Inland Feeder Project (MWD 1998), SBKR surveys for SBVWCD (Dames & Moore 1999), SBKR and

general biological surveys conducted for ACOE and SBVWCD (URS 2000 through 2003), and Dudek site visits in 2004 and 2006.

2.0 ENVIRONMENTAL SETTING

2.1 Existing Land Use

Existing land uses within the Plan Area include aggregate mining and processing operations, ground water recharge, maintenance and operation of flood control facilities, utility maintenance (e.g., power corridor maintenance, well facility maintenance), and roadways (*Figure 3*).

- ◆ Aggregate mining and processing are conducted by Cemex Construction Materials, L.P. (Cemex), Robertson's Ready Mix (Robertson's), and Matich on a total of 857 acres permitted in the western half of the project area.
- ◆ The SBVWCD maintains water spreading basins on approximately 64 acres in the eastern section of the project area.
- ◆ The San Bernardino County Flood Control District (SBCFCD) maintains flood control facilities on approximately 350 acres along the Santa Ana River, Plunge Creek, and City Creek (LSA 2005).
- ◆ The Metropolitan Water District of Southern California (MWD) utility easement (approximately 36 acres) lies within the general boundaries of the Wash Planning Area.
- ◆ Roadways (and utility easements within their right-of-way), utility easements maintained by Southern California Edison (SCE), and a right-of-way for the Atchison Topeka and Santa Fe Railroad (LSA 2004).
- ◆ Private roadways in the Wash Planning Area include private mining company haul roads, SBVWCD roads to access basins and canals, and SBCFCD service roads on top of levees. These roadways will be maintained in place; with the exception of mining roads to be addressed in Section 3.2 below (Covered Activities).
- ◆ The U.S. Army Corps of Engineers (ACOE), and the Orange, Riverside, and San Bernardino County Flood Control Districts created the approximately 760-acre Woolly-star Preserve Area (WSPA) for the state- and federally-listed endangered Santa Ana River woollystar along the Santa Ana River in the southern part of the Wash Planning Area through consultation between the ACOE and the Service for the construction of the Seven Oaks Dam (Chambers Group, Inc. 1993).

The BLM owns and manages four parcels of public lands totaling 1,040 acres within the planning area. Under the South Coast Resource Management Plan (1994), three of the parcels totaling 760 acres were designated as the Santa Ana River ACEC and RNA.

The ACEC and RNA are managed for the protection of the Santa Ana River woollystar and slender-horned spineflower. The remaining public lands are managed for multiple-uses while maximizing protection of sensitive resources. Under the Recreation and Public Purpose Act, the Inland Fish and Game Club operates and maintains a shooting range on approximately 20 acres of land on BLM land adjacent to Orange Street and the Southern California Edison power-line corridor in the north-west section of the Plan Area; this shooting range is not a part of the proposed project.

2.2 Physical Characteristics

The project site is located in the broad fluvial plain formed by the deposition of the Santa Ana River, Mill Creek, and City Creek as they flow southwest from the San Bernardino Mountains. Several fault bounded structural blocks saddle the general site area. The down dropped San Bernardino Valley block underlies the site and represents a buried rift between the San Andreas Fault to the northeast, and the San Jacinto Fault to the southwest. As the block subsided, alluvium derived from the San Bernardino Mountains filled the resulting depression, causing a maximum alluvial thickness of 600 to 1,200 feet east of the San Bernardino International Airport. It is this alluvium that is mined throughout the Wash Plan. The alluvial deposit is of the Quaternary Age and consists of igneous and metamorphic clasts whose rocks are found in the mountains and at Crafton Hills. The clasts' sizes vary from that of fine size to boulders in size. All materials on the project site are classified in the Soboba Series, specifically Soboba Stony loamy sand.

The site is subject to ground shaking from earthquakes but is not located within an Alquist-Priolo special studies zone. The area is generally level and is not subject to landslide hazards. Depth to ground water fluctuates with season and groundwater recharge activities. The area is subject to liquefaction though this is not considered hazardous for mine or reclamation, recharge, and flood control activities.

Climate - The San Bernardino Valley is characterized by a climate of long dry summers and short wet winters. Annual average daily temperatures range from a low of 49° F. to an average high of 80° F. The average rainfall is about 15.6" per year, with approximately 90 percent falling from November through March.

Ground Water - The project site overlies the Bunker Hill Ground Water Basin. The Bunker Hill Basin is one of the largest ground water basins in the Santa Ana River Basin and is a ground water recharge zone. This basin, whose boundaries are generally defined by earthquake faults, which effectively act as subsurface dams trapping ground water, is bounded on the north and east by the San Bernardino Mountains, on the southeast by the Crafton Hills and the Badlands, and on the west by the San Jacinto fault. Because faults can act as barriers to the movement of ground water, the faults in the vicinity of the SBVWCD Mill Creek recharge facilities may restrict the movement of water into the larger Bunker Hill basin. Three subareas within the Bunker Hill Basin have been identified. These are commonly referred to as Bunker Hill I, Bunker Hill II,

and the Pressure Zone. The project site overlies the Bunker Hill II subarea. The Pressure Zone to the west is an area where high ground water levels have historically existed.

Many natural and artificial phenomena such as rainfall, natural stream inflow, evaporation, ground water extractions through wells, and spreading operations for replenishment of the water supply influence ground water levels in the Bunker Hill Basin. The Bunker Hill Basin is artificially recharged by several agencies. Included are surface stream diversions made for ground water replenishment by the SBVWCD on the Santa Ana River and Mill Creek, and facilities operated by the SBCFCD on Devil Creek, Twin Creek, Waterman Creek, and Sand Creek, which may also be used for ground water recharge. The SBVWCD and its predecessors have been diverting water from the Santa Ana River and Mill Creek for over 90 years.

Soils - The Santa Ana River extends the length of the project area; two tributaries to the Santa Ana River also occur within the project area, Plunge Creek in the north and Mill Creek in the southeast. Soils within the project area are mapped as Soboba stony loamy sand, 2 to 9% slopes, Psamments and Fluvents, frequently flooded, and Hanford coarse sandy loam, 2 to 9% slopes (*Figure 4*). Soils in and along the channels of the Mill Creek, the Santa Ana River, Plunge Creek, and an old channel between Plunge Creek and the Santa Ana River (roughly 15% of the project area) are mapped as Fluvents and Psamments. These are recent soils with little or no evidence of horizon development. Fluvents are formed by recent water-deposited sediments in floodplains, fans, and stream or river deltas and consist of layers of various soil textures. Psamments formed on terraces or outwash plains and contain well sorted, freely draining soils that always contain sand, fine sand, loamy sand or coarse sand in subsoils between 10 and 40 inches depth.

Most of the project area consists of Soboba stony loamy sand. This soil forms on alluvial fans in granitic alluvium and typically contains stony loamy sand, very stony loamy sand, and very stony sand to a depth of approximately 60 inches. Included within this soil are areas of Tujunga gravelly loamy sand.

A small area of Hanford coarse sandy loam occurs in the northeastern part of the project area. This is a well-drained soil formed in recent granitic alluvium on valley floors and alluvial fans that contains sandy loam to a depth of about 60 inches.

2.3 Vegetation and Land Cover

Seven vegetation and land cover were mapped onsite: variations of Riversidean alluvial fan sage scrub, Riversidean sage scrub, chamise chaparral, non-native grassland, disturbed habitat, open water and developed land (*Figure 5*).

2.3.1 Riversidean Alluvial Fan Sage Scrub

Riversidean alluvial fan sage scrub is a Mediterranean shrubland type that occurs in washes and on gently sloping alluvial fans. Alluvial scrub is made up predominantly of drought-deciduous soft-leaved shrubs, but with significant cover of larger perennial species typically found in chaparral (Kirkpatrick and Hutchinson 1977). Scalebroom (*Lepidospartum squamatum*) generally is regarded as an indicator of Riversidean alluvial scrub (Smith 1980; Hanes *et al.* 1989). In addition to scalebroom, alluvial scrub typically is composed of white sage (*Salvia apiana*), spiny-leaved redberry (*Rhanmus crocea*), buckwheat (*Eriogonum* spp.), our Lord's candle (*Yucca whipplei*), California croton, cholla (*Opuntia* spp.), tarragon (*Artemisia dracuncululus*), yerba santa (*Eriodictyon* spp.), mule fat and mountain-mahogany (*Cercocarpus betuloides*) (Hanes *et al.* 1989; Smith 1980).

Riversidean alluvial fan sage scrub occurs on alluvial benches throughout the Wash Planning Area, in various stages of succession. During various field studies conducted from 2000 to 2003, URS had mapped pioneer, intermediate and mature Riversidean alluvial fan sage scrub within the project area. The three stages of succession generally represent the differences in species composition, growth forms (*i.e.*, woodiness of plants) and percent cover. More mature areas tend to have woodier vegetation, higher percent cover and greater diversity than younger areas.

Areas mapped as mature Riversidean sage scrub are typically those areas most distant from human disturbances (*e.g.*, recharge basins, roads, mining pits, etc.) and the main flows of the Santa Ana River, Plunge Creek and Mill Creek. The vegetation consists of woody shrubs and fully developed subshrubs and physical characteristics include fine silty soils with few cobbles. Typical species include California juniper (*Juniperus californica*), chamise (*Adenostoma fasciculatum*), our Lord's candle, spiny redberry, holly-leaved redberry, hoaryleaf ceanothus (*Ceanothus crassifolius*) and sugarbush (*Rhus ovata*) (URS October 26, 2003).

Areas mapped as intermediate Riversidean sage scrub typically lie between mature and pioneer Riversidean sage scrub. The vegetation is fairly dense and consists primarily of subshrubs. Physical characteristics include coarse and fine sands with cobbles. Typical species include California buckwheat (*Eriogonum californica*), prickly pear cactus (*Opuntia phaeantha*), deerweed, yerba santa (*Eriodictyon trichocalyx* var. *trichocalyx*), and our Lord's candle (URS October 26, 2000)

Areas mapped as intermediate/mature Riversidean sage scrub exhibit physical and vegetative characteristics found in both intermediate and mature Riversidean sage scrub (URS October 26, 2003).

Areas mapped as pioneer Riversidean sage scrub are generally located adjacent to human disturbances and along the Santa Ana River, Plunge Creek and Mill Creek where scouring and sediment deposits result in changing substrates. The vegetation is typically sparse, of low stature and low diversity. Physical characteristics consist of boulders and cobbles without top soil. Typical species include deerweed, California buckwheat, scalebroom, and mulefat (*Baccharis salicifolia*) (URS October 26, 2003).

2.3.2 Riversidean Sage Scrub

Riversidean sage scrub is dominated by a characteristic suite of low-statured, aromatic, drought-deciduous shrubs and subshrub species. It is a more xeric expression of coastal sage scrub, occurring further inland in drier areas where moisture and climate are not moderated by proximity to the marine environment. Riversidean sage scrub typically occurs on steep slopes, severely drained soils or clays that are slow to release stored soil moisture (Holland 1986). Species composition varies substantially depending on physical circumstances and the successional status of the habitat; however, characteristic species include California sagebrush (*Artemisia californica*), buckwheat (*Eriogonum* spp.), laurel sumac (*Malosma laurina*), California encelia (*Encelia californica*), and several species of sage (e.g., *Salvia mellifera*, *S. apiana*) (Holland 1986). Other common species include brittlebush (*E. farinosa*), lemonadeberry (*Rhus integrifolia*), sugarbush, yellow bush penstemon (*Keckiella antirrhinoides*), Mexican elderberry (*Sambucus mexicanus*), sweetbush (*Bebbia juncea*), boxthorn (*Lycium* spp.), coastal prickly-pear (*Opuntia littoralis*), coastal cholla (*O. prolifer*), tall prickly-pear (*O. oricola*), and species of dudleya (*Dudleya* spp.).

Onsite, Riversidean sage scrub includes brittlebush, deerweed (*Lotus scoparius*), spiny redberry, California sagebrush, California buckwheat, white sage (*Salvia apiana*) and laurel sumac. Physical characteristics include gravely, sandy and/or silty soil with few cobbles (URS October 26, 2003).

2.3.3 Chamise Chaparral

Chamise chaparral occurs throughout much of the range of chaparral in California from approximately 30 to 6000 feet in elevation. This vegetation is found on all slope-aspects generally on shallow soils and is dominated by chamise (*Adenostoma fasciculatum*). Vegetation structure is open to dense from approximately 3 to 13 feet in height, with little litter and few understory species in mature stands (URS October 26, 2000). Onsite this vegetation type is dominated by chamise but also includes yerba santa, California buckwheat, sugar bush, our Lord's candle with an understory of non-native brome grasses and gracile buckwheat (*Eriogonum gracile*).

2.3.4 Non-native Grassland

Disturbance by maintenance (e.g., mowing, scraping, discing, spraying, etc.), grazing, repetitive fire, agriculture, or other mechanical disruption may alter soils and remove native seed sources from areas formerly supporting native habitat. Within the Wash Planning Area, non-native grassland consists of a sparse to dense cover of annual grasses as well as native and non-native annual forb species. Physical characteristics include clay soils or fine-textured loamy soils (URS October 26, 2003).

2.3.5 Disturbed Habitat

Disturbed habitat refers to areas that lack vegetation entirely but do not contain an impermeable surface. These areas are generally the result of severe or repeated mechanical perturbation. Onsite, these areas are characterized by weedy, introduced annuals, including black mustard (*Brassica nigra*), telegraph weed (*Heterotheca grandiflora*), red-stemmed filaree (*Erodium cicutarium*), and non-native grasses such as bromes (*Bromus madritensis* and *B. diandrus*) and wild oat (*Avena barbata*) (URS October 26, 2003).

2.3.6 Open Water

The recharge basins were constructed onsite by the SBVWCD. These basins contain standing water intermittently during the year. When dry, they can be characterized as similar to disturbed habitat described above.

2.3.7 Developed Land

Developed land refers primarily to mining pits and paved roads throughout the project area. However, developed land also includes previously graded areas, landscaped areas and areas actively maintained or utilized in association with existing developments.

2.4 Wildlife

Based on a review of biological surveys prepared for the 400-acre Robertson's Ready Mix site (Lilburn Corporation 1996) and the 630-acre Sunwest Materials' site (Lilburn Corporation 1997), both within the Wash Planning Area, seventy-seven wildlife species were observed or detected. These species included three amphibians, 17 mammals, 11 reptiles and 46 birds. The bird species include a variety of upland birds, such as mourning dove (*Zenaida macroura*), killdeer (*Charadrius vociferus*), Say's phoebe (*Sayornis saya*), scrub jay (*Aphelocoma coerulescens*), and house finch (*Carpodacus mexicanus*). Raptors include American kestrel (*Falco sparverius*), white-tailed kite (*Elanus caeruleus*), and red-tailed hawk (*Buteo jamaicensis*). Amphibians included

western toad (*Bufo boreas*), Pacific treefrog (*Pseudacris regilla*) and western spadefoot. Observed mammals include striped skunk (*Mephitis mephitis*), coyote (*Canis latrans*), California ground squirrel (*Spermophilus beecheyi*), Virginia opossum (*Didelphis virginiana*), and desert cottontail (*Sylvilagus audubonii*). In addition, the California side-blotched lizard (*Uta stansburiana elegans*), western fence lizard (*Sceloporus occidentalis longipes*), and silvery legless lizard (*Anniella pulchra pulchra*) were observed.

3.0 COVERED SPECIES

3.1 Listed Species

3.1.1 Santa Ana River Woollystar

SPECIES DESCRIPTION

Status

Santa Ana River woollystar was federally-listed as endangered on September 28, 1987 (52 Federal Register 36265) and state-listed as endangered in January 1987. The species is on the California Native Plant Society's List 1B with an R-E-D code of 3-3-3. Critical habitat has not been designated for the Santa Ana River woollystar.

Habitat and Habitat Associations

The Santa Ana River woollystar is found only within open washes and early-successional alluvial fan scrub on open slopes above main watercourses on fluvial deposits where flooding and scouring occur at a frequency that allows the persistence of open shrublands. Suitable habitat is comprised of a patchy distribution of gravelly soils, sandy soils, rock mounds and boulder fields (Zembal and Kramer 1984; Zembal and Kramer 1985; U.S. Fish and Wildlife Service 1986). Suitable habitat typically contains low amounts of clay, silt and micro-organic materials (Burk *et al.* 1989). These areas typically maintain a perennial plant cover of less than 50%. Associated perennial plants include California buckwheat (*Eriogonum fasciculatum*), California croton (*Croton californicus*), yerba santa (*Eriodictyon trichocalyx*) and scale-broom (*Lepidospartum squamatum*) (Burk, *et al.* 1989; Zembal and Kramer 1984; Zembal and Kramer 1985). The Santa Ana River woollystar is an early-successional species and possibly requires flood-mediated habitat rejuvenation (Wheeler and Burk 1990). Sheet flood flows probably occur in this habitat every one hundred to two hundred years (U.S. Fish and Wildlife Service 1986). A 1989 study of woollystar habitats and surrounding habitats

revealed that the % cover of European annuals is lowest in woollystar habitats (Burk, *et al.* 1989).

Biology

Genetics: The Santa Ana River woollystar is one of five subspecies of the perennial sub-shrub *Eriastrum densifolium*. This species exhibits complex morphological variation and the subspecies are difficult to distinguish (Brunell and Whitkus 1993). Based on a study of nineteen quantitative characters, the single morphological variation within *Eriastrum densifolium* is corolla tube length: the Santa Ana River woolly star has a significantly longer corolla than the other four subspecies (Brunell and Whitkus 1999a). The results of an analysis of cross-compatibility confirmed that the species is self-incompatible but indicates that each of the subspecies is compatible with the other subspecies. Viable seeds were produced by 54 pair-wise crosses of individuals from 24 populations, representing all five subspecies (Brunell and Whitkus 1999b).

A genetic study using Random Amplified Polymorphic DNA (RAPD) markers indicates no major discontinuity between the Santa Ana River woollystar and other subspecies: Santa Ana River woollystar is not a member of a distinct genetic population group. RAPD marker variation within *E. densifolium* is a continuum and differentiation among populations is related to geographic distance not morphological characteristics. The subspecies appear to be polyphletic: to have developed from more than one ancestral type (Brunell and Whitkus 1994; Brunell and Whitkus 1997). On the subspecies level, an enzyme electrophoresis study indicated genetic variation within *E. densifolium* ssp. *sanctorum* is higher than expected for an endemic taxon. The degree of allozyme diversity indicates the lack of inbreeding or recent population bottlenecks; this subspecies is not as genetically vulnerable as other endemic taxa (Brunell and Rieseberg 1993).

Reproduction: Santa Ana River woollystar blooms from June to August (Munz 1974). This obligate outcrosser has bright lavender-blue flowers that occur in heads of about twenty large (over one and a quarter inches long) blossoms (Burk *et al.* 1989). Pollen release occurs before the stigma of the same flower becomes receptive so pollen gatherers are unlikely pollinators. According to field observations by Burk *et al.* (1989), of the eight insect families and a hummingbird observed visiting woollystar, only digger bees, an anise swallowtail butterfly, a hummingbird and the giant flower-loving fly are capable of reaching the woollystar flower. The giant flower-loving fly (*Raphiomidas actoni* ssp. *actoni*) was found to be the most abundant pollinator at one Santa Ana River woollystar population (Burk *et al.* 1989). The peak abundance of the giant flower-loving fly correlates with Santa Ana River woollystar flowering and the flower-loving fly depends on sandy substrate for reproduction. These two species may be mutually dependent at some locations (Burk *et al.* 1989). At other population sites, the digger

bee (*Micranthophora flavocincta*) or hummingbirds (including the black-chinned hummingbird, *Archilochus alexandri*) were observed to be the most abundant pollinator (Muñoz 1991 as cited in Jigour and Roberts 1996; Erickson 1993; Chambers 1993).

Scarification of seeds is not necessary and the optimum germination temperature is approximately 60° Fahrenheit. Leaching by one inch of simulated rainfall significantly increases germination as compared to wetted seeds. Seed viability is high: up to 99% (Burk *et al.* 1989). Germination follows early winter rains; however, many of the seedlings die in the following spring and summer (Chambers 1993).

Dispersal: A study by Burk *et al.* (1989) during the 1986-87 growing season revealed that 900 to 1000 seeds were produced per plant and 92% fell within one foot of the parent plant. Sixty inches (five feet) was the longest dispersal distance observed. Woollystar outer seed coats form a mucilaginous (sticky) mass that binds the seed to surrounding soil particles. Therefore, longer dispersal distances probably are associated with flood events (Burk *et al.* 1989; Jigour and Roberts 1996).

Demography: Studies have shown that Santa Ana River woollystar seedlings are more successful in recently flooded habitat (Wheeler and Burk 1990; Burk *et al.* 1989). Average life span for the woolly-star individual is five years and the oldest individual observed was ten years (Burk *et al.* 1989).

Known Distribution

The Santa Ana River woollystar occurs from about 150 to 580 meters above mean sea level (AMSL) along the Santa Ana River and Lytle and Cajon Creek flood plains from the base of the San Bernardino Mountains in San Bernardino County southwest along the Santa Ana River through Riverside County into the Santa Ana Canyon of northeastern Orange County (Munz 1974; Patterson 1993; Roberts 1998; Zembal and Kramer 1985; Patterson and Tanowitz 1989).

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum, the UCR herbarium and available literature. There are 950 occurrences of Santa Ana River woollystar recorded in the Wash Planning Area; one occurrence was from the CNDDDB, four occurrences were from the UCR herbarium, 49 occurrences were from the Robertson's Ready Mix Project reports (Lilburn Corporation 1996) and 896 occurrences were from the Sunwest Materials Project reports (Lilburn Corporation 1997). The species occurs along the floodplain of the Santa Ana River, Plunge Creek and Mill Creek. Of the 946 occurrences, 41 were mapped in developed areas and may no longer be extant. The remaining occurrences were mapped in

Riversidean sage scrub; pioneer, intermediate and mature Riversidean alluvial fan sage scrub; disturbed habitat and the recharge basins.

Special Biological Considerations

This species is a low shrubby perennial that blooms from June to August. There are three primary pollinators: long-tongued digger bee, giant flower-loving fly and hummingbirds. The importance of a particular pollinator type appears to depend on habitat type within the floodplain. Due to high seedling mortality rates, this life stage is the most critical for survival of the species (Chambers 1993). This species is associated with early- to moderate- successional alluvial scrub, and thus requires periodic flooding and silting for the creation of new habitats and colonization. Sustaining Santa Ana River woollystar will require maintaining a portion of the Santa Ana River alluvial floodplain and possibly artificially reestablishing protected habitats adjacent to aging woollystar populations for future colonization (Burk *et al.* 1989).

Threats to Species

This species is threatened by floodplain modification for flood control purposes and development; flood control management (clearing for channel maintenance and construction of flood control structures); off-road vehicle activity; grazing (resulting in heavy weed cover); farming; sand and gravel mining; and loss of habitat and competition with aggressive non-native species such as European grasses and river cane (*Arundo donax*) (Zemba and Kramer 1985; Burk *et al.* 1989; U.S. Fish and Wildlife Service 1986).

3.1.2 Slender-horned Spineflower

SPECIES DESCRIPTION

Status

Slender-horned spineflower was federally-listed as endangered on September 28, 1987 (52 Federal Register 36265) and state-listed as endangered in January 1982. The species is on the California Native Plant Society's List 1B with an R-E-D code of 3-3-3. Critical habitat has not been designated for the slender-horned spineflower.

Habitat and Habitat Associations

At the majority of sites, slender-horned spineflower is found in sandy soil in association with mature alluvial scrub (Reveal and Hardham 1989; Rey-Vizgirdes 1994). In the Vail

Lake area in Riverside County this species is also associated with gravel soils of Temecula arkose deposits in association with open chamise chaparral (Boyd and Banks 1995; Gordon-Reedy 1997). Prigge, *et al.* (1993) found that the ideal habitat appears to be a terrace or bench that receives overbank deposits every 50 to 100 years.

Cryptogamic crusts are frequently present in areas occupied by slender-horned spineflower (Boyd and Banks 1995; U.S. Fish and Wildlife Service 1986). These crusts on the soil surface are composed of associations of bryophytes (mosses), algae, lichens, and some xerophytic liverworts (Harper and Marble 1988 as cited in U.S. Fish and Wildlife Service 1996). Cryptogamic crusts enable soils to retain moisture and may help suppress invasion by non-native plant species (Boyd and Banks 1995; U.S. Fish and Wildlife Service 1996).

Biology

Genetics: Slender-horned spineflower was described as *Centrostegia leptotheca* by Goodman (1934 as cited in Reveal and Hardham 1989); however, morphological characters and cytological studies indicate that this species is unique. Therefore, slender-horned spineflower has been placed in a monospecific genus (Reveal and Hardham 1989).

Although this species is protandrous (anthers develop earlier than the stigma), suggesting that slender-horned spineflower is an obligate outcrosser (U.S. Fish and Wildlife Service 1996), Reveal (1989 as cited in Prigge, *et al.* 1993) determined that slender-horned spineflower is self-compatible. Because the population sizes are large enough (hundreds to thousands of individuals), it is doubtful that this species is experiencing a genetic bottleneck (Reveal 1989 as cited in Prigge, *et al.* 1993).

Reproduction: This herbaceous annual blooms from April through June and has white to pink flowers (1.2 to 2 mm in length). The flowers produce small (1.7 to 2 mm long), brown or black achenes (Reveal and Hardham 1989). Because slender-horned spineflower is an annual and a spring-bloomer, it is expected to germinate following winter precipitation (Prigge, *et al.* 1993).

Dispersal: The involucre of the slender-horned spineflower has six ascending awns and six descending awns, suggesting that this species is ideally suited for animal dispersal. Potential dispersal agents include coyotes, rabbits, rodents and deer. Dispersal may also occur via flood water or wind (Prigge, *et al.* 1993; U.S. Fish and Wildlife Service 1996).

Demography: Slender-horned spineflower has a spreading habit and is sparsely glandular (Reveal and Hardham 1989). This annual is small and prostrate, with heights of 5 to 15 cm and diameters of 3 to 10 cm (U.S. Fish and Wildlife Service 1986).

Known Distribution

Slender-horned spineflower is endemic to southwestern cismontane California, ranging from central Los Angeles County east to San Bernardino County, and south to southwestern Riverside County in the foothills of the Transverse and Peninsular Ranges, at 200 to 700 meters elevation (Hickman 1993). Only eight areas are still known to support slender-horned spineflower, including two localities each in Los Angeles County (Bee Canyon and Big Tijuana Wash), and two in San Bernardino County (the Santa Ana River Wash and Cajon Wash) (Reveal and Hardham 1989; Rey-Vizgirdes 1994; CNDDDB 2000). There are four areas known to support slender-horned spineflower in western Riverside County: Temescal Wash, upper San Jacinto River, and the north flank of Agua Tibia Mountains (at Arroyo Seco and Kolb Creek) (Prigge, *et al.* 1993; CNDDDB 2000, Rey-Vizgirdes 1994; Gordon-Reedy 1997; Banks 1999).

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum, the UCR herbarium and available literature. There are 44 occurrences of slender-horned spineflower recorded in the Wash Planning Area; seven occurrences were from the CNDDDB, one occurrence was from the UCR herbarium, one occurrence was from the Robertson's Ready Mix Project reports (Lilburn Corporation 1996) and 35 occurrences were from the Sunwest Materials Project reports (Lilburn Corporation 1997). The species occurs along the floodplain of the Santa Ana River and Plunge Creek. Of the 44 occurrences, three were mapped in disturbed areas. The remaining 41 occurrences were mapped in intermediate and mature Riversidean alluvial fan sage scrub.

Special Biological Considerations

This species is generally dependent on mature alluvial scrub that is maintained by periodic flooding and sediment transport. It is important for the survival of this species that adequate alluvial scrub habitat and active fluvial processes be maintained. Individuals are small, and thus may be difficult to locate. This species is only readily detectable in the spring between April and June when in bloom. Population size varies considerably from year to year depending upon rainfall.

Threats to Species

This species is threatened by urbanization, off-road vehicle use, sand and gravel mining, trampling associated with recreation, flood control measures (*i.e.*, constriction of the floodplain, dams, etc.), and competition from non-native plant species (U.S. Fish and Wildlife Service 1986; Prigge, *et al.* 1993).

3.1.3 Coastal California Gnatcatcher

SPECIES DESCRIPTION

Status

The coastal California gnatcatcher was federally-listed as threatened on March 25, 1993 (50 CFR Part 17 RIN 1018-AB56). The species is not listed as threatened or endangered by the state of California, but is designated a Special Concern species. Critical habitat was proposed for the species on April 24, 2003 (50 CFR Part 17 RIN 1018-AI72). Critical habitat has not yet been made final. The proposed critical habitat area does not include any portion of the Wash Planning Area. The California gnatcatcher is also on the United States Bird Conservation Watch List and the Audubon Watch List.

Habitat and Habitat Associations

The coastal California gnatcatcher (gnatcatcher), a subspecies of the California gnatcatcher, is a small member of the thrush family (Muscicapidae). The gnatcatcher typically occurs in or near sage scrub habitat, which is a broad category of vegetation that includes the following plant communities as classified by Holland (1986): Venturan coastal sage scrub, Diegan coastal sage scrub, maritime succulent scrub, Riversidean sage scrub, Riversidean alluvial fan sage scrub, southern coastal bluff scrub, and coastal sage-chaparral scrub. Coastal sage scrub is composed of relatively low-growing, dry-season deciduous and succulent plants. Characteristic plants of this community include California sagebrush (*Artemisia californica*), various species of sage (*Salvia* sp.), California buckwheat (*Eriogonum fasciculatum*), lemonadeberry (*Rhus integrifolia*), California encelia (*Encelia californica*), and *Opuntia* spp. Ninety-nine percent of all gnatcatcher locality records occur at or below an elevation of 984 feet (Atwood 1990).

Coastal sage scrub is patchily distributed throughout the range of the gnatcatcher, and the gnatcatcher is not uniformly distributed within the structurally and floristically variable coastal sage scrub community. Rather, the subspecies tends to occur most frequently within the California sagebrush-dominated stands on mesas, gently sloping areas, and along the lower slopes of the coastal ranges (Atwood 1990). An analysis of the percent gap in shrub canopy supports the general impression that gnatcatchers

prefer relatively open stands of coastal sage scrub (Bontrager 1991). The gnatcatcher occurs in high frequencies and densities in scrub with an open or broken canopy while it is absent from scrub dominated by tall shrubs and occurs in low frequencies and densities in low scrub with a closed canopy (Weaver 1998). The territory size increases as vegetation density decreases and with distance from the coast, probably due to food resource availability. Thus, gnatcatchers will use even sparsely vegetated coastal sage scrub for shelter and to forage for insects as long as perennial shrubs are available (ERCE 1990).

Gnatcatchers also use chaparral, grassland, and riparian or alluvial habitats where they occur adjacent to sage scrub (Bontrager 1991). The use of these habitats appears to be most frequent during late summer, autumn, and winter, with smaller numbers of birds using such areas during the breeding season. These non-sage scrub habitats are used for dispersal, but data on dispersal use are largely anecdotal (Bowler 1995; Campbell *et al.* 1995). Although existing quantitative data may reveal relatively little about gnatcatcher use of these other habitats, these areas may be critical during certain times of the year for dispersal or as foraging areas during drought conditions (Campbell *et al.* 1998). Breeding territories have also been documented in non-sage scrub habitat. Campbell *et al.* (1998) discuss likely hypotheses explaining why non-CSS habitat is used by gnatcatchers including food source availability, dispersal areas for juveniles, temperature extremes, fire avoidance, and lowered predation rate for fledglings.

Environmental, vegetational, and food-abundance characteristics are important aspects of territory quality; however, they are related to the time of year when the evaluation is made (Redak *et al.* 1997). Based on the studies of Redak *et al.* (1997) during the breeding season, habitat use was negatively associated with distance to the coast and the elevation of the territory. The habitat use was positively associated with the abundance of adult stages of beetles, flies, spiders and larval stages of all arthropods. Plots with high densities of California sagebrush, flat-topped buckwheat, and white sage were also used by birds. In contrast, during the non-breeding season, the correlation of habitat use with vegetation and location variables remained but the correlation was no longer present with the invertebrate communities.

Biology

Genetics: The coastal California gnatcatcher was originally described as a distinct species by Brewster (1881) based on specimens, however, Grinnell (1926) concluded that it was a subspecies of the black-tailed gnatcatcher (*Polioptila melanura*) which is widely distributed throughout the Sonoran and Chihuahuan deserts of the southwestern United States and Mexico. Atwood (1980, 1988) concluded that the species was specifically distinct from *P. melanura*, based on differences in ecology and behavior, which was adopted by the American Ornithologists' Union Committee on Classification

and Nomenclature (American Ornithologists Union 1957, 1989, 2003). Recent mitochondrial DNA sequencing confirmed the species-level recognition of the Coastal California gnatcatcher, which was calculated to differ from the black-tailed gnatcatcher (*P. melanura*) by 4.0%, similar to differences calculated in the black-capped gnatcatcher (*P. nigriceps*) and white-lored gnatcatcher (*P. albiloris*) (Zink and Blackwell 1998).

Diet and Foraging: The coastal California gnatcatcher is primarily insectivorous, non-migratory, and exhibits strong site tenacity (Atwood 1990). The diet deduced from fecal samples resulted in leaf- and plant hoppers and spiders predominating the samples. True bugs, wasps, bees, and ants were only minor components of the diet (Burger *et al.* 1999). Gnatcatcher adults selected prey to feed their young that was larger than expected given the distribution of arthropod size available in their environment, and chicks were provisioned with larger prey items and significantly more grasshoppers and crickets and spiders. Both adults and young consumed more sessile than active prey items (Burger *et al.* 1999).

The richness of the insect community within a habitat area may be a useful tool for describing the quality of the habitat (Burger *et al.* 1996). This is especially important for strictly insectivorous species such as the coastal California gnatcatcher. Gnatcatcher habitat use has been positively associated with total insect species richness and total individual insect abundance (Redak *et al.* 1996). Thus overall food abundance and diversity plays an important role in territory selection and use for this species (Redak *et al.* 1996). Habitat use during the non-breeding season showed no clear relationship to any component of the invertebrate community (Redak *et al.* 1997).

Daily Activity: Activity budget data indicate that gnatcatchers are most active and vocal during the morning. A lull in activity usually occurs during mid-day and activity increases again late in the day (Mock *et al.*, 1990).

Reproduction: The breeding season of the gnatcatcher extends from mid February through mid-August, with the peak of nesting activity occurring from mid-March through mid-May. The gnatcatcher nest is a small, cup-shaped basket usually found one to three feet above the ground in a small shrub or cactus. Clutch sizes range between three and five eggs, with the average being four. Juvenile birds associate with their parents for several weeks (sometimes months) after fledging (Atwood 1990). The coastal California gnatcatcher is a year-round resident. Nest building begins during the mid part of March with the earliest recorded egg date approximately March 20 (Mock *et al.*, 1990). Post-breeding dispersal of fledglings occurs between late May and late November. Predation may be a major source of nest failure (Bontrager 1991; Grishaver *et al.* 1998).

Nest site attendance by male gnatcatchers was determined to be equal to that of females for the first nest attempt and then decline to almost 1/3 of that of the female for later nesting attempts (Sackman 1998).

The frequency with which various plant species have been recorded as nesting substrata indicates the overall preference of the sage scrub community as the habitat type (Atwood 1980). California sagebrush was chosen 25% of the time with other species including white sage, black sage, chamise, cholla, buckthorn, orange, lemonadeberry, and others making up the balance of nest shrub selections (Atwood 1980).

Survival: Gnatcatchers are persistent nest builders and often attempt multiple broods typically upon nesting failure, which is suggestive of a high reproductive potential. This is, however, typically offset by high rates of nest predation and brood parasitism (Atwood 1990). High rates of nest failure may account for the high number of nesting attempts of the coastal California gnatcatcher (Grishaver *et al* 1998). In western Riverside County, 78.9% of the nesting attempts failed with 52.9% suffering from nest predation (Braden 1999). Gnatcatchers typically live for two to three years, although ages of up to five years have been recorded for some banded birds (Braden *et al.* 1995). Most of the juvenile birds usually die during the cold winter months, although the percentage was not quantified. Observations indicate that gnatcatchers are highly vulnerable to extreme cold, wet weather (Mock *et al.*, 1990).

Dispersal: Dispersal is a means by which genetic and demographic exchange between subpopulations maintains the viability of the regional metapopulation (Bailey and Mock 1998). Details regarding the dispersal effect on genetic and demographic connectivity of subpopulations and the actual requirements for dispersal are largely unknown (Rotenberry and Scott 1998) but some information can be documented from anecdotal observations. The mean dispersal distance of gnatcatchers banded as nestlings for males was 2.85 km and for females was 3.33 km (Atwood *et al.* 1996). Mean dispersal of juveniles in Orange County was found to be 1.05 km with one individual dispersing a total of 7.55 km (Galvin 1998). Although the mean dispersal distances that have been documented above are relatively low, dispersal of juveniles is difficult to observe and to document without extensive banding studies. It is likely that the few current studies underestimate the gnatcatcher's typical dispersal capacity because of the difficulty of detecting (Bailey and Mock 1998). Juvenile coastal California gnatcatchers are apparently able to traverse highly man-modified landscapes, including non-native landscaping vegetation, for at least short distances and this underestimation of the species' dispersal capability can lead to an overestimation of the metapopulation's vulnerability to extinction (Bailey and Mock 1998). A few observations of gnatcatcher dispersal behavior indicate that a stepping stone linkage, that is, a series of small patches of suitable habitat interspersed with developed habitat, is

deemed acceptable for situations where the habitat is otherwise fragmented and no contiguous linkage is available (Bailey and Mock, 1998). Additionally, natural and restored coastal sage scrub habitat along highway corridors has been documented to be used for foraging and nesting by gnatcatchers and may serve important dispersal functions (Famolaro and Newman 1998). Typically, however, the dispersal of juveniles requires a corridor of native vegetation which provides foraging and cover opportunities to link larger patches of appropriate sage scrub vegetation (Soule 1991). These dispersal corridors may facilitate the exchange of genetic material and provide a path for recolonization of areas from which the species has been extirpated and may provide increased mating opportunities for unpaired birds (Soule 1991; Galvin 1998).

Socio-Spatial Behavior: The coastal California gnatcatcher seems to become highly territorial by late February or early March each year. Males seem to be very vocal during this time period (Mock *et al.*, 1990). In San Diego County the territory size for inland sites was calculated to range between 13 and 39 acres per pair, averaging 24 acres per pair (ERCE 1990). In Riverside County, it was estimated that about 24 acres of sage scrub habitat was required per pair of coastal California gnatcatchers (Braden 1998, pers. comm.). The distribution of the gnatcatcher is thought to be related to elevation with most of the birds located below 250 m elevation within 35 km of the coast and 500 meter elevation for inland regions (Atwood and Bolsinger 1992). During the non-breeding season, gnatcatchers have been observed to wander in adjacent territories and unoccupied habitat increasing their home range size to approximately 78% larger than their breeding territory (Preston *et al.* 1998). Estimates of the territory size should be examined with caution as the calculation may be influenced by differences in data collection and analysis (Atwood *et al.* 1998).

Coastal California gnatcatchers are most often observed in pairs even in the non-breeding season. They appear to maintain their territories and are relatively sedentary throughout the year (Dunn and Garrett 1987). In fact vocalization rates, which may provide communication within the pair, were highest from August through March (Preston *et al.* 1998).

Community Relationships: Predation occurs in greater proportion in the upper and lower one third of the nest shrub. Predation was lower in nests with full clutch sizes which may indicate the parents are more attentive to the nest after the clutch is complete (Sockman 1997). Potential predators include scrub jays, greater roadrunners, and cactus wrens which have been observed to be actively mobbed by the gnatcatcher (Bontrager 1991). The coastal California gnatcatcher also is known to be affected by nest parasitism of the brown-headed cowbird. However, the gains in nest success from decreased nest parasitism appear to be negated by increased nest abandonment due to predation before cowbirds have migrated into an area (Braden *et al.* 1997). Thus, although a cowbird trapping program may reduce parasitism significantly and lower

abandonment due to parasitism, nest predation then increases and negates the benefit of the trapping program (Braden *et al.* 1997). Nest parasitism apparently has resulted in earlier nesting dates of the gnatcatcher which may help compensate for the negative affect of parasitism (Patten and Campbell 1998).

Although the coastal California gnatcatcher may serve as an adequate “umbrella species” for other species that occur in similar habitats and that require a similar territory size or smaller (Fleury *et al.* 1998), it is not a particularly good indicator of bird-species richness in coastal sage scrub habitat (Chase *et al.* 1998).

Known Distribution

Historically, the coastal California gnatcatcher occurred from southern Ventura County southward through Los Angeles, Orange, Riverside, San Bernardino, and San Diego counties, and into Baja California, Mexico, to approximately 30 degrees north latitude near El Rosario (Atwood 1990). A detailed analysis of elevational limits associated with gnatcatcher locality records reveals that a significant portion, 65 to 70% of the historic range, may have been located in southern California rather than Baja California (USFWS 2000). The gnatcatcher was considered locally common in the mid-1940's, but by the 1960's this subspecies had declined substantially in the United States owing to widespread destruction of its habitat (Atwood 1990). Currently, the subspecies occurs on coastal slopes of southern California, ranging from southern Ventura southward through Palos Verdes Peninsula in Los Angeles County through Orange, Riverside, San Bernardino and San Diego Counties into Baja California to El Rosario, Mexico, at about 30 degrees north latitude (Atwood 1991). In 1993, the USFWS estimated that approximately 2,562 pairs of gnatcatchers remained in the United States. Of these, 30 pairs occurred in Los Angeles County, 757 pairs occurred in Orange County, 261 pairs occurred in Riverside County, and 1,514 pairs occurred in San Diego County.

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum and available literature. There are two occurrences of coastal California gnatcatcher in the Wash Planning Area; both occurrences are found in the San Bernardino County Natural History Museum database and the USFWS database. One occurrence is located within mature Riversidean alluvial fan sage scrub in a land use area currently designated as flood plain. The other occurrence is located within pioneer Riversidean alluvial fan sage scrub in a land use area currently designated as water ways. Both locations are in the central portion of the Wash Planning Area.

Special Biological Considerations

Knowledge of the demography of a population is fundamental to determining long-term trends. For birds, the demographic parameters of primary importance are annual breeding success, defined as number of chicks fledged per pair, and recruitment, defined as percentage of fledglings that enter the breeding population. A preliminary demographic model for a population of coastal California gnatcatchers based on observed data was prepared by Woehler *et al* (1995). Based on the results, for the population to be stable, each breeding pair must replace themselves over their lifetime. The data from a population at U.C. Irvine had a 90% mortality (that is, a 10% recruitment) and produced 0.64 fledglings per egg. In western Riverside County, 78.9% of the nesting attempts failed which translates to an approximately 80% mortality (Braden 1999). Thus, for the U.C Irvine example, a pair must produce 30 eggs during their lifetime and must live for five years for the population to remain stable (Woehler *et al*. 1995).

Gnatcatcher populations appear to be inversely correlated to seasonal total rainfall (Erickson and Miner 1998). Thus increased rainfall during the winter is a mixed blessing in that it is likely that it increases winter mortality but may increase the productivity of the invertebrate prey population base (Erickson and Miner 1998). Other weather-related factors that may influence the distribution of the gnatcatcher include the January mean minimum temperature which, for the coastal California gnatcatcher, has been estimated to be approximately 2.5 degrees C (Mock 1998). This metabolic constraint may preclude gnatcatchers, as a sedentary bird, from occupying otherwise suitable habitat within their range (Mock 1998). This link between a species distribution and physiological adaptations to the climate has been shown previously for other bird species (Hayworth and Weathers 1984).

Gnatcatchers nested earlier, had more successful nests, produced more fledglings, had a longer nesting period, and had lower fledgling costs when their territories were associated with increased grass and forb cover, increased perennial structure, increased horizontal perennial homogeneity, decreased vertical perennial homogeneity, and decreased perennial diversity (Braden *et al*. 1997). Thus, assessment of habitat quality should take into account these variables for preserve planning.

A habitat-based metapopulation model developed for the coastal California gnatcatcher revealed that the model predicted a fast decline and high risk of population extinction with most combinations of population parameters. The results were most sensitive to density-dependent effects, the probability of weather-related catastrophes, adult survival, and adult fecundity (Akçakaya and Atwood 1997). However, this metapopulation model resulted in the greatest difference on a time horizon of only a few decades. This may be appropriate if the model is used to compare alternative management options but not to make assessments with longer time horizons (Akçakaya

and Atwood 1997). Studies providing information on long-scale demography and metapopulation variables are still largely unknown (Rotenberry and Scott 1998).

Structure of the perennial vegetation within coastal sage scrub seems to be an important component leading to successful gnatcatcher reproduction (Braden, 1997). Any disturbance that affects perennial structure and homogeneity within gnatcatcher territories, such as fire or grazing, also may affect gnatcatcher fitness (Braden, 1997). In general, recently burned areas are not used by gnatcatchers except on an occasional basis, and five to seven years of recovery may be necessary before gnatcatchers will nest in burned areas (Atwood *et al.* 1998, Beyers *et al.* 1994). This slow recovery of coastal sage scrub may be due to invasion of exotic annuals after any disturbance and may also be affected by invasion of exotic annuals, as well as air pollution (O'Leary and Westman 1988; O'Leary (1990). The frequency of fires in wildland areas tends to increase as fragmentation increases due to urbanization and agricultural activity.

Fire may be an important factor to consider in the conservation design for this species and management plans may be necessary to provide a strategic framework for merging the needs for the species with the challenges of fire control (Mackey *et al.* 1994). Consideration of habitat refugia, burn frequency, and recolonization of recovering burn areas will be necessary for designing reserves for this species (Atwood, *et al.* 1998). Observations after a major fire of coastal sage scrub reveal that a large proportion of the gnatcatchers within the burned area were displaced to adjacent habitat rather than killed outright and were packed more densely into remaining areas of intact coastal sage scrub (Atwood, *et al.* 1998). On unburned areas within San Diego and Riverside counties, two pairs per hectare were found, but only 0.02 pair per hectare were found on burned areas (Mayer and Wirtz 1995). However, the gnatcatchers that are able to establish territories on burned areas appear to breed at rates very similar to those on unburned sites (Wirtz and Mayer 1995). The extent and timing of vegetation recovery may determine the habitat suitability for breeding pairs (Wirtz and Mayer 1995).

The presence of gnatcatchers within burned areas may indicate post-fire dispersal or the availability of refugia from the fire (Mayer and Wirtz 1995). Frequent burning of coastal sage scrub may lead to domination of the site by introduced grasses, in addition, burned sage scrub often remains unsuitable for breeding gnatcatchers for a number of years after burning (ERCE 1991, Zedler *et al.* 1983, O'Leary 1990). At an inland site, burned 12 years earlier, there was less than 10% shrub cover and no gnatcatchers, while annual grasses and mustard species dominated the site (Beyers *et al.* 1994). Management consideration will need to take into account that large scale fires may damage gnatcatcher populations in both the burned area and the refugia area (Atwood *et al.* 1998). Duplicate linkages also provide for the temporary loss of function of the coastal sage scrub habitat in the event of a fire (Campbell *et al.* 1998). A duplicity of habitat linkages composed of sage scrub and other undeveloped habitats which may

be composed of native as well as non-native habitats may provide for dispersal ability of the species (Campbell *et al.* 1998).

Other factors that will be relevant for designing a reserve system for the gnatcatcher will be the dispersal distance and average territory size. In western Riverside County, the average dispersal distance for juvenile gnatcatchers has been documented as 1.14 km (Braden *et al.*, 1994a). The distances may be influenced by many factors such as sex, reproductive opportunities, available habitat and other factors. The average territory size for gnatcatchers is 8.42 acres during the breeding season and can expand to 60 acres during the non-breeding season (Braden and Powell, 1994b). A reserve design for this species will need to maintain connections of breeding habitat such that dispersal between areas can be accomplished and that are large enough to accommodate the largest territory sizes.

Linkages of habitat along linear features such as highways and power-line corridors may be of significant value in linking populations of the gnatcatcher (Famolaro and Newman 1998). Stepping stone linkages which are designed to function as habitat linkages are acceptable but should be line of sight as much as possible (Bailey and Mock 1998). The width of a linkage is recommended to be approximately 1,200 feet. This will provide a linkage wide enough to support a gnatcatcher territory. For linkages less than this width, the gnatcatchers currently mapped for occurring within the linkage may not be able to remain within the area. In the case of narrow habitat widths, the linkage will serve the function of connection of habitat areas only. An important linkage of habitat to maintain or encourage is that from the western Riverside County area north into San Bernardino County (Davis *et al.* 1998). This linkage is within the Jurupa Hills and connects to the Santa Ana River in Riverside County (Davis *et al.* 1998). Recently, gnatcatchers have been observed within Jurupa Hills although the area is not surveyed regularly (Davis *et al.* 1998).

The coastal California gnatcatcher may be suitably analyzed on a landscape or habitat basis rather than on a data point basis due to the fact that it responds well to habitat management and will readily occupy revegetated coastal sage scrub (O'Connell and Erickson 1998, Miner *et al.* 1998). The fact that gnatcatchers will occupy revegetated coastal sage scrub makes this an important component of long-term management for the species (O'Connell and Erickson 1998). In one study, restored habitat was included in 19 of 22 gnatcatcher territories and nests in restored areas were as likely to produce at least one young as nests in naturally generated scrub (Miner *et al.* 1998). Additionally, in this study, thirteen percent of the nests were placed within 3 meters of actively used roads or trails and their success rate was similar to that of nests placed further from these high-use areas, thus breeding success of gnatcatchers appears not to be negatively affected by current management practices and levels of public use

within park areas. (Miner *et al.* 1998). Given that this study was conducted within specific State Park areas, additional studies would be helpful for other situations.

The continued fragmentation of habitat over time has increased exposure of gnatcatcher to threats associated with habitat edge (Atwood 1993). Numerous nest predators thrive on habitat edges, and brood parasitism by the brown-headed cowbird (*Molothrus ater*) appears to be exacerbated by increased edge effects (Bolger *et al.* 1997, Atwood 1993). Management of edge effects of future development may also be needed although there is little evidence that coastal California gnatcatchers are negatively affected by having their territory located at the edge of urban development (Atwood, 1998). This may be more associated with the use of the gnatcatcher of sage scrub/grassland ecotone. In support of this observation, studies of “edge/fragmentation reduced” species versus “edge/ fragmentation enhanced” species places the coastal California gnatcatcher in an “edge/fragmentation insensitive” category (Bolger *et al.* 1997). This is a category occupied by the characteristic species of shrub habitats in the region. They tend to be abundant and widely distributed across the landscape and habitat gradients although their abundance is much lower than the other species in the group (Bolger *et al.* 1997). Other forms of edge management may still be required. Management may include fencing areas occupied by gnatcatchers to protect birds from human and other intruders. Management may include exotic plant removal along edges of development or planting with native shrubs. Analysis of the current shrub cover to shrub cover and composition in 1934 indicates a drastic reduction in native species and increase in non-native grasses and forbs (Minnich and Dezzani 1998).

Threats to Species

In 1997, the total number of gnatcatchers in the United States was estimated at 2,899 pairs, after subtracting out all gnatcatcher pairs authorized for Take under Habitat Loss Permits, approved natural Community Conservation Plans, Habitat Conservation Plans, and section 7 consultations (“Reinitiation of formal consultation on implementation of the special rule for the coastal Coastal California gnatcatcher [1-6-93-FW-37R1]”). This apparent increase in abundance since 1993 is likely the result of additional surveys occurring within previously unsurveyed areas, as well as increased productivity in response to favorable climatic conditions (USFWS 2000).

Although observed declines in numbers and distribution of the gnatcatcher resulted from numerous factors, habitat destruction, fragmentation and adverse modification are the principal reasons for the gnatcatcher's current threatened status (USFWS 1993). The amount of coastal sage scrub available to gnatcatchers has continued to decrease during the period after the listing of the species. It is estimated that up to 90% of coastal sage scrub vegetation has been lost as a result of development and land conversion (Westman 1981a, 1981b; Barbour and Major 1977), and coastal sage scrub is

considered to be one of the most depleted habitat types in the United States (Kirkpatrick and Hutchinson 1977; Axelrod 1978; Klopatek *et al.* 1979, Westman 1987; O'Leary 1990). The fragmentation of habitat may artificially increase populations in adjacent preserved habitat; however, these population surpluses may be lost in subsequent years due to crowding and lack of resources (Scott 1993). In addition, agricultural use, such as grazing and field crops, urbanization, air pollution, increases in fire frequency and the introduction of exotics have all had an adverse impact on extant sage scrub habitat. A consequence of urbanization that is contributing to the loss, degradation, and fragmentation of coastal sage scrub is an increase in wildfires due to anthropogenic ignitions (human caused fires). High fire frequencies and the lag period associated with recovery of the vegetation may significantly reduce the viability of affected subpopulations of the gnatcatcher (USFWS 1991).

3.1.4 San Bernardino Kangaroo Rat

SPECIES DESCRIPTION

Status

The San Bernardino kangaroo rat was emergency federally-listed as endangered on January 27, 1998 (63 Federal Register 3835) and a final rule determining the San Bernardino kangaroo rat to be endangered was published on September 24, 1998 (63 Federal Register 51005). A proposed rule for designation of critical habitat for the San Bernardino kangaroo rat was published on December 8, 2000 (65 Federal Register 77178) and the final rule for critical habitat was published on April 23, 2002 (67 Federal Register 19812). The San Bernardino kangaroo rat is a California Special Concern Species.

Habitat and Habitat Associations

The following habitat description draws heavily upon the San Bernardino kangaroo rat habitat assessment conducted by URS (2003a) for the San Bernardino Valley Water Conservation District.

The San Bernardino kangaroo rat, a subspecies of the Merriam's kangaroo rat (*Dipodomys merriami*), typically is found in Riversidean alluvial fan sage scrub and sandy loam soils, alluvial fans and flood plains, and along washes with nearby sage scrub (McKernan 1997 as cited in USFWS 1998). Braden and McKernan (2000) suggest that the San Bernardino kangaroo rat also occurs in other habitats in their range, including chaparral and even disturbed areas that are associated with alluvial processes.

Riversidean alluvial fan sage scrub vegetation within the Santa Ana River floodplain is comprised of three primary seral stages of alluvial fan sage scrub: pioneer, intermediate, and mature phases.

Pioneer phase alluvial fan sage scrub is the initial colonizing stage where recent scouring and flood events have occurred. This phase is characterized by very sparse distributions of subshrubs dominated by bristly goldenaster (*Heterotheca sessiflora* spp. *echioides*) and scalebroom (*Lepidospartum squamatum*). Because of typically recent scouring and flooding and the lack of a well-developed vegetation community, pioneer phase alluvial fan sage scrub is less suitable for the San Bernardino kangaroo rat compared to intermediate alluvial fan sage scrub. URS (2003) argued that even though San Bernardino kangaroo rats may be captured in pioneer alluvial fan sage scrub during trapping programs, it should be considered “the least likely place” for San Bernardino kangaroo rat because of the lack of habitat requirements suitable for establishing burrow systems. However, these areas are integral to the overall habitat system and life history of the San Bernardino kangaroo rat with regard to temporary use and dispersal and potential succession to more suitable habitat over time.

Intermediate phase alluvial fan sage scrub is comprised mainly of subshrubs such as California buckwheat (*Eriogonum fasciculatum*), brittlebush (*Encelia farinosa*), yerba santa (*Eriodictyon tricholax*), Spanish bayonet (*Yucca whipplei*), deerweed (*Lotus scoparius*), valley cholla (*Opuntia parryi*), and coastal prickly-pear (*Opuntia littoralis*). Intermediate phase alluvial fan sage scrub typically occur on terraces above scoured channels. Intermediate phase alluvial fan sage scrub is considered the highest quality habitat for the San Bernardino kangaroo rat because this phase retains open, sandy areas favored by the species. Intermediate phase alluvial fan sage scrub is expected to support the highest densities of the San Bernardino kangaroo rat of the three primary seral stages.

Mature phase alluvial fan sage scrub typically occurs on higher terraces away from the active flood channel that have not been subjected to flooding and scouring for many years. The mature phase is characterized by large woody species such as California juniper (*Juniper californicus*), Spanish bayonet, chamise (*Adenostoma fasciculatum*), holly-leaved cherry (*Prunus ilicifolia*), sugarbush (*Rhus ovata*), redberry (*Rhamnus crocea*), hoaryleaf ceanothus (*Ceanothus crassifolius*). Because the mature phase is characterized by relatively dense vegetation with few sandy openings, it is considered less suitable for the San Bernardino kangaroo rat than the intermediate phase. However, as with the pioneer stage, San Bernardino kangaroo rats may occasionally use mature alluvial fan sage scrub (as well as adjacent upland areas) and it may serve an important refugia function during large flood events (USFWS 2000).

The San Bernardino kangaroo rat typically occupies sandy loam substrates that allow for the digging of simple, shallow burrows (McKernan 1997 as cited by USFWS 1998). *D. merriami*, and other kangaroo rat species, actively avoid rocky substrates (Brown and Harney 1993). The large majority of the Plan Area supports Soboba stony loam sand, with a substantial component of psamments (sands) and fluvents within the historic active channels (NRCS Soil Survey Geographic [SSURGO] Database 2004). There are smaller pockets of Cienega sandy loam and Hanford coarse sandy loam. All these soils are suitable for the San Bernardino kangaroo rat either as burrowing habitats, where the soils have been consolidated by vegetation as occurs in intermediate alluvial fan sage scrub, or as foraging habitat in pioneer alluvial fan sage scrub.

Biology

There are few specific studies of the subspecies San Bernardino kangaroo rat, but there is a substantial literature for the species *D. merriami*. The information presented in this section largely is for the species, with specific reference to the San Bernardino kangaroo rat where appropriate.

Genetics: Williams *et al.* (1993) provides descriptions for 19 subspecies of *D. merriami*. Patton and Rogers (1993a, 1993b) provide reviews of what is known of the cytogenetics (*e.g.*, chromosomal variation) and biochemical genetics (*e.g.*, isozyme and allozyme analyses, DNA sequencing) of heteromyid rodents, the rodent family to which *D. merriami* belongs. Patton and Rogers generally conclude that the understanding of heteromyid genetics is still relatively poor, the data are uneven, and that few studies have applied recent technical developments (*e.g.*, DNA fingerprinting and sequencing). As of 1993, the only biochemical technique applied to heteromyids is protein electrophoresis, a relatively crude analytic tool by today's standards. Of interest to conservation planning would be any information relating genetics to habitat fragmentation and isolation, demography, habitat tolerance, and speciation. Unfortunately, very little information in the literature is available to address these issues.

D. merriami has 52 chromosomes and there is no reported karyotypic variation in the species (Patton and Rogers 1993a). The proportion of gene loci that are polymorphic among individuals ranges from 0.06 to 0.16 and the mean proportion of loci that are heterozygotic within individuals ranges from 0.00 to 0.061. These values, as well as values for other kangaroo rat species, are relatively low compared to other mammals (Patton and Rogers 1993b). (Patton and Rogers [1993b] caution that these summary statistics probably contain large sampling error as well as other important sources of error that limit their interpretation. Also, protein electrophoresis cannot provide the fine-grain genetic analysis possible with DNA fingerprinting and other recent techniques.) Studies of electromorphic distance for *D. merriami* also indicate high degrees of genetic

similarity. Although Lidicker (1960) remarked that the San Bernardino kangaroo rat was noticeably smaller and more differentiated compared to other *D. merriami*, there is no existing evidence that it is genetically distinct from other subspecies. Furthermore, there are no genetic studies of different populations of the San Bernardino kangaroo rat to address the effects of habitat fragmentation and isolation, demography, or other issues relevant to conservation planning.

The only genetic demographic study of *D. merriami* identified by Patton and Rogers (1993b) was a study of spatial relationships among individual genotypes in a population of *D. merriami* on a 10-acre study site near Kramer, California by Johnson and Selander (1971). This study concluded, in Patton and Rogers' words, "that spatial clustering of genotypes was evident at two loci, and suggested that local structure, including the possibility of inbreeding, may characterize local kangaroo rat populations." page 264. However, their findings did not include statistical corroboration of this finding and these results must be interpreted as very preliminary.

Diet and Foraging: Many studies have reported on the diet of *D. merriami* (see Reichman and Price 1993 for a comprehensive review), but no specific studies have been conducted on the San Bernardino kangaroo rat. Nonetheless, it is unlikely that the San Bernardino kangaroo rat exhibits meaningfully different feeding patterns compared to other subspecies of *D. merriami* that would be relevant for conservation planning. *D. merriami* are primarily granivores (seed eaters), but they ingest herbaceous material and insects when available (Bradley and Mauer 1971; Reichman and Price 1993). They collect seeds from the substrate into fur-lined cheek pouches for transport and then store them in scattered surface caches in the vicinity of their home burrows for later retrieval and consumption (Daly *et al.* 1992a). Unlike some larger kangaroo rat species (*e.g.*, *D. spectabilis*), *D. merriami* do not hoard seeds to a central location (*i.e.*, larder hoarding). Bipedal locomotion in kangaroo rats allows them to travel large distances over open ground very quickly and exploit widely scattered food sources.

Daily Activities: *D. merriami*, and all other kangaroo rats, are primarily nocturnal animals, but they also exhibit crepuscular behavior around dusk and dawn. They emerge from their day burrows around dusk to engage in foraging and other activities. Animals may be active any hour of the night, but the heaviest concentration of activity tends to occur in the three- to four-hour time span just after dusk. They usually return permanently to their day burrows before dawn (Behrends *et al.* 1986a). Factors affecting the amount and patterns of surface activity of individuals include: (1) sex and reproductive condition, with reproductive active males traveling farther than female or males with regressed testes (Behrends *et al.* 1996a); and (2) moonlight, with animals reducing surface activity and shifting activity toward places with relatively dense cover (Lockard and Owings 1974; Price *et al.* 1984). Daly *et al.* (1992b) found that *D. merriami* shifted from nocturnal activity during full moon to more crepuscular activity

during dawn and dusk periods, suggesting a more complex and fine-grain compensatory behavioral response to moonlight rather than simply reducing overall surface activity to avoid moonlight.

Reproduction: The species *D. merriami*, and heteromyids in general, have relatively low reproductive output for rodents (see Wilson *et al.* 1985). In the wild, *D. merriami* and other kangaroo rat species typically breed one or two times per year, with the peak breeding being mid-winter through spring, although they may breed more frequently in good years (Duke 1944; Fitch 1948; Quay 1953; Pfieffer 1956; Holdenreid 1957; Reynolds 1960; Beatley 1969; Bradley and Mauer 1971, 1973; Kenagy 1973; Reichman and Van De Graaf 1973, 1975; Van De Graaff and Balda 1973; Flake 1974). Field observations of reproductive activity by *D. merriami* include several records of females producing successive litters at intervals of about two months, with a minimum interval of about 45-50 days (Daly *et al.* 1984). Breeding activities appear to vary in relation to ecological conditions, and individuals may not breed in years when conditions are poor. In good years, females are known to breed in their natal season (Daly *et al.* 1984). Studies indicate that nearly all adult individuals in a population are capable of breeding, but the proportion of individuals active at non-peak breeding periods (*e.g.*, late summer-early fall) may be smaller (*e.g.*, Kenagy 1973). Fall and winter rains, and the consequent production of herbaceous annuals, appear to be an important factor for breeding activities, but the positive effects do not always occur in the following season; *i.e.*, there may be lag effects in the correlation between rainfall, production of herbaceous annuals, and kangaroo rat reproduction (*e.g.*, Beatley 1969; Chew and Butterworth 1964). Herbaceous vegetation is ingested in greater quantities during the breeding season (Bradley and Mauer 1973; Reichman and Van De Graaff 1975), and there is experimental evidence that herbaceous material or free water is necessary for successful reproduction (Soholt 1977).

A captive breeding study of *D. merriami* by Daly *et al.* (1984) found that mean litter size for 129 deliveries of captive bred females was 2.4, with few litters exceeding four pups. Interestingly, 10 litters of wild-conceived litters averaged 3.7 pups. The modal gestation period for *D. merriami* in this study was 33 days. *D. merriami* do not have a post-partum estrus (*i.e.*, receptivity in conjunction with parturition), but they may become reproductively active within four days of removal of a nursing litter. Pups appear to stop nursing at about 25 days. The youngest mother in this captive breeding study conceived at 64 days of age and gave birth at 97 days. In the field, a female conceived her first litter between 40 and 50 days (Daly *et al.* 1984). *D. merriami* exhibit clear estrous cycles with a median length of 13.4 days and spontaneous ovulation (Wilson *et al.* 1985).

Based on field and laboratory studies of *D. merriami*, the maximal annual reproductive output of an individual female, based on a typical litter of two or three pups, is unlikely to

exceed ten (Wilson *et al.* 1985), which is far below many other rodents that exhibit induced ovulation or post-partum estrous (*e.g.*, murids).

Survival: Individual *D. merriami* have observed life spans of at least five years in the wild and at least seven years in captivity (Behrends, pers. obs.; Daly *et al.* 1990). However, the data on expected life span and annual survivorship of *D. merriami* in the field are equivocal because of the many practical limitations in measuring and interpreting survivorship (*e.g.*, distinguishing between mortality and emigration). Nonetheless, French *et al.* (1967) estimated a life expectancy for *D. merriami* of 4.3 months in the Mojave Desert. Chew and Butterworth (1964) observed 12-19% annual survivorship in a trapping study in the Mojave Desert, with most disappearances occurring from October to April and attributable to juvenile disappearances and the harsh winter. Zeng and Brown (1987), on the other hand, concluded that adult survivorship appears to be relatively high and year-to-year survivorship of males and females appears to be very similar. Because *D. merriami* are long-lived and recruitment of juveniles into populations probably varies from year-to-year, most populations are comprised primarily of adults. After correcting for emigration, annual adult survivorship may be on the order of 75% (Brown and Harney 1993).

In a long-term study of predation of a *D. merriami* population in Palm Desert, California, Daly *et al.* (1990) recorded a total of 50 known or presumed predations and found that more mobile individuals were at higher risk of predation; general survivorship was not estimated because of the lack of control for emigration. Important predators in the Daly study were coyotes, snakes, owls, and shrikes. Bobcats and foxes also would be expected to be important predators of the San Bernardino kangaroo rat in western Riverside County.

Dispersal: Jones (1989) determined that *D. merriami* is philopatric; *i.e.*, individuals tend to establish home ranges in proximity to their natal range. Dispersal in *D. merriami* is slightly male-biased, but more than 85% of individuals disperse less than 125 meters over their lifetimes (Jones 1989). Although recruitment of juveniles into the population is unknown, it probably varies in relation to breeding activities and ecological conditions (*i.e.*, carrying capacity of the habitat). The data collected by French *et al.* (1967) and Chew and Butterworth (1964) suggests that juveniles are at high risk of disappearance, either through dispersal or mortality.

Socio-Spatial Behavior: Radio-telemetry studies and live-trapping studies of *D. merriami* have elucidated the basic patterns of this species' social and spatial behavior (*e.g.*, Behrends *et al.* 1986a,b; Jones 1989). A review of heteromyid behavioral adaptations by Randall (1993) summarizes the fundamental aspects of *D. merriami* social organization. Although day burrows tend to be dispersed, this species exhibits overlapping home ranges. However, female-female overlap is less than male-male and

male-female range overlap. Individuals primarily are solitary and asocial, although aggressive and non-aggressive interactions are not rare and individuals tend to tolerate familiar neighbors more than strangers. Core areas around day burrows may be aggressively defended. Although home ranges shift spatially over time, individuals tend to have long-term associations with the same individuals. Average home ranges of males and females are similar in size, and range from 0.16 ha (0.4 acre) in Arizona to 2.6 ha (6.4 acres) in Texas, with individual home ranges varying substantially (Behrends *et al.* 1986b).

That kangaroo rats are relatively long-lived (> 7 years in captivity), exhibit conservative reproductive traits, juvenile mortality exceeds adult mortality (French *et al.* 1967; Zeng and Brown 1987) and individuals disperse little between birth and adulthood (Jones 1989) all suggest that *D. merriami* has long-term stability in social communities.

Population densities of *D. merriami* can vary dramatically, probably in association with resource availability, but tempered by the conservative life history traits of the species; *i.e.*, relatively low fecundity and recruitment of juveniles, storage of seeds, and effective predator avoidance. Geographically, typical population densities are variable and range from lows of 1 individual/ha in Texas to about 18 individuals/ha in Arizona (Behrends 1986b; Brown and Harney 1993). Typical densities in the Palm Desert area of California were approximately 6 individuals/ha over a five-year period (Behrends, pers. obs.). Subsequent trapping studies demonstrated an enormous range in abundance; fewer than 10 individuals were trapped on a 1-ha grid in drought years and more than 80 individuals in years following substantial rainfall and high production of food resources (Behrends, pers. obs.) (note that these are not density estimates for a unit area because the 1-hectare grid draws animals from beyond the grid). Reynolds (1958) conducted a 12-year trapping study in southern Arizona and recorded densities of 3.4 individuals/ha to a high of 17.3 individuals/ha. Zeng and Brown (1987) recorded population densities ranging between about 2 and 18 individuals/ha in the Chihuahuan Desert in southeastern Arizona.

Community Relationships: The community ecology of heteromyid rodents, including kangaroo rats (*Dipodomys* spp.), pocket mice (*Perognathus* and *Chaetodipus* spp.) and kangaroo mice (*Microdipodops* spp.) is among the most studied aspect of this family's biology. Brown and Harney (1993) provide a comprehensive overview and attempted synthesis of this complex subject. Presented here are some generalizations that fall from this large body of literature.

Arid grassland and desert environments support a surprising diversity of coexisting rodent granivores. The diversity and number of coexisting species vary depending on local conditions and the requirements of the constituent species. For example, the San Bernardino kangaroo rat potentially overlaps with two other kangaroo rats (*D. stephensi*

and *D. simulans*), at least two pocket mice (*Chaetodipus fallax* and *Perognathus longimembris*), and at least four murids (*Peromyscus maniculatus*, *P. eremicus*, *Neotoma lepida*, and *Reithrodontomys megalotis*) that would compete for space and food resources. Brown and Harney (1993) conclude that “the composition of these assemblages is not random. Instead it is determined by interactions of the species with the physical environment, with other kinds of organisms, and with other rodent species.” page 646. Generally, species that do coexist tend to occupy and exploit different microhabitats or niches or differ in their seasonality of resource exploitation. For example, a trapping program conducted along Wilson Creek east of Sage in Riverside County, California recorded three species of kangaroo rats: *D. merriami collinus*, *D. stephensi* and *D. simulans*. *D. merriami* was trapped in coarse, sandy soils adjacent to the creek, *D. stephensi* was trapped in sparse grassland and a dirt road away from the creek, and *D. simulans* was trapped in coastal sage scrub on the slopes above the creek (Dudek 1995).

D. merriami exhibits somewhat greater habitat tolerance than other heteromyids. A survey of community assemblages by Brown and Harney (1993) found that *D. merriami* has one of the broadest geographic ranges and tends to be one of the most abundant species of assemblage where found.

Interspecific competition is an important component of the organization of heteromyid community structure. For example, competitive exclusion can result in nonrandom assemblages that partition the resources and habitats in the community. Other potential mechanisms of resource partitioning listed by Brown and Harney (1993) include habitat selection or restriction, independent adaptations, food partitioning and variable foraging efficiency, seed distribution, resource variability, predator-mediated coexistence, aggressive interference, and seasonality.

Kangaroo rats and other heteromyid rodents also modify their environments (Brown and Harney 1993). They dig burrows, which moves the soils and provides habitat and refugia for other species, including other rodents, reptiles, amphibians, birds and invertebrates. Collection, storage and consumption of seeds by kangaroo rats have profound effects on the vegetation structure of the habitats they occupy. For example, experiments by Brown and his colleagues in southeastern Arizona have demonstrated that kangaroo rats are a “keystone guild” where their removal from plots resulted in the habitat converting from desert shrub to grassland (Brown and Heske 1990). In addition, resource use by kangaroo rats substantially overlaps with that of seed-eating birds and harvester ants. Where kangaroo rats have been excluded in experimental plots, ants have increased dramatically (Brown and Harney 1993).

The coevolutionary results of such inter- and intraspecific community relationships and their relationship to plant communities are not understood, but it can be concluded that

rodents are an important component of arid ecosystems. In addition to their direct impacts on plant communities, they are important prey for a variety of predators and their presence also affects populations of other prey such as small reptiles, lagomorphs and some birds (Brown and Harney 1993).

Physiological Ecology: Kangaroo rats and most other heteromyid species live in arid environments characterized by hot summers, long, cold winters, unpredictable precipitation, and ephemeral primary productivity of food sources (French 1993). For example, *D. merriami* has been observed on the surface at temperatures of -19 degrees Celsius (Kenagy 1993). Living in such extreme environmental conditions has high metabolic and thermoregulatory costs.

Kangaroo rats are perhaps most famous for their water conservation capabilities. Schmidt-Nielsen (1964) and French (1993) summarized the behavioral and physiological means by which kangaroo rats, and *D. merriami*, in particular, conserve water: they occupy burrows during daylight hours to avoid high temperatures; their evaporative water loss is much lower than other mammals when corrected for body mass; they have relatively low metabolic rates (about 30% lower than average mammals); they produce low volumes of highly concentrated urine and low-moisture feces; and their water requirements can be satisfied by oxidative or metabolic water in conjunction with the seeds and herbaceous material they consume. *D. merriami* also produces highly concentrated milk, thus minimizing lactational water loss.

Energy conservation is very important for species living in extreme environments. *D. merriami* is active on the surface the entire year (e.g., Behrends *et al.* 1986b, Kenagy 1973). Other than at times of starvation, there is no evidence that *D. merriami* goes into torpor (a kind of hibernation) to conserve resources, as do pocket mice (*Perognathus* and *Chaetodipus*) and kangaroo mice (*Microdipodops*) (French 1993). However, *D. merriami* does tend to rest at temperatures at the lower end of thermal neutrality whenever possible to conserve energy (French 1993).

These physiological and behavioral characteristics allow kangaroo rats to inhabit a broad range of arid habitats in western North America, as well as allow individuals to survive during long periods of adverse climatic conditions.

Known Distribution

According to Hall (1981), the species *D. merriami* occupies a broad range of grasslands and arid habitats in southwestern North America, extending from northwestern Nevada southward through southeastern California, Baja California and in mainland Mexico south to northern Sinaloa. It ranges eastward to southeastern Utah, western and southern Arizona, central and southern New Mexico, and into western Texas.

The historic range of the subspecies San Bernardino kangaroo rat lies west of the desert divide of the San Jacinto and San Bernardino mountains and extends from the San Bernardino Valley in San Bernardino County to the Meniffee Valley in Riverside County (Lidicker 1960; Hall 1981). The USFWS estimates that at the time of listing in 1998, the San Bernardino kangaroo rat occupied approximately 6,576 ha (16, 440 acres) of suitable habitat in about seven general locations (USFWS 2000), including the Santa Ana River, Cajon Creek Wash, Lytle Creek Wash, City Creek, and upper Etiwanda Wash in San Bernardino County, and San Jacinto River and Bautista Creek in Riverside County.

Status in Wash Planning Area

Various habitat assessments and associated trapping studies by URS (1999, 2000a-d, 2003a-d) and the San Bernardino County Museum have consistently found the San Bernardino kangaroo rat in suitable habitat throughout the Wash Planning Area, with 62 discrete mapped locations. Because the discrete occurrences only indicate trap lines where the species has been trapped, however, they should not be considered the extent of occupied or suitable habitat. Because of the mobility of this species and its opportunistic use of habitat and fluctuations in habitat suitability related to environmental conditions and events (e.g., floods, drought), virtually any suitable habitat could be expected to be used by the San Bernardino kangaroo rat at some point.

Based on existing land use patterns, 38 occurrences (61%) are located in the flood plain, 12 (20%) are located in the waterways, 7 (11%) are located in the mining area, 4 (6%) in the water conservation area, and 1 (2%) in the basin. Thus 81% of the occurrences are located in areas expected to have the highest habitat suitability: the floodplain and waterways.

With regard to vegetation types, the distribution of occurrences is consistent with the characterization of habitat suitability of the three seral phases of alluvial fan sage scrub; i.e., pioneer, intermediate and mature. Based on the existing vegetation map, 12 occurrences (20%) occur in the pioneer phase, 31 (51%) occur in the intermediate (24 occurrences) and intermediate-mature phase (seven occurrences), and only five (8%) occur in the mature phase (including one occurrence in mature/non-native grassland). Only one occurrence each is located in basin and chaparral/non-native grassland. However, seven occurrences (11%) are located in developed and five (8%) are located in disturbed areas. However, these data cannot be used to develop a habitat suitability index because the sampling regimes used in the trapping programs were biased towards trapping in the most suitable habitat areas. Nonetheless the data are consistent with the known habitat associations of this species.

Special Biological Considerations

Maintaining an adequate amount of suitable habitat to accommodate stochastic events (flooding, drought, habitat succession) will be important for this species in the Wash Planning Area. San Bernardino kangaroo rats experience fluctuations in habitat quality based on the fluvial processes tied to flooding events and drought. Intermediate alluvial fan sage scrub, which occurs on terraces between pioneer and mature habitats, probably provides the best habitat for the species because it does not flood often, but also is fairly open (7-22% cover) with a low shrub canopy. The density of vegetation is particularly important for kangaroo rats as it affects their burrowing, locomotion and foraging ability. The experimental removal of vegetation can result in an increase in kangaroo rats using the more open habitat (Rosenzweig 1973; Price 1978). Pioneer and mature sage scrub stages, on the other hand, are less suitable; pioneer areas are subject to frequent flooding and mature alluvial scrub may become too dense in cover for this species. Consequently, natural fluvial processes, whereby cycles of flooding and dry periods result in dynamic fluctuations of habitat, probably are crucial for this species.

Threats to Species

Habitat Loss: Identified threats to the San Bernardino kangaroo rat include the loss of habitat, habitat fragmentation, urban and industrial development, highway construction, flood control and water conservation projects, sand and gravel mining, grazing, and vandalism (USFWS 1998). Additional threats to the species likely include farming and disking of habitat for weed abatement, heavy grazing, and off-road vehicles. Although this species is associated with sandy washes and drainages, permanent habitat supporting sparse alluvial fan sage scrub and other occupied habitat (e.g., Riversidean sage scrub, chaparral, grasslands and disturbed habitat) often may not be in areas under the jurisdiction of the U.S. Army Corps of Engineers (i.e., within the ordinary high water mark of the drainage) or California Department of Fish and Game (i.e., streams with bed and bank). For example, non-jurisdictional benches above creek channels probably are important for this species.

Genetic Isolation: Although there appears to be little genetic variation in kangaroo rats in general (Patton and Rogers 1993a,b), a study by Johnson and Selander (1971) suggested some degree of local genetic structure and the possibility inbreeding in a population *D. merriami* in Kramer, California. With such small and currently isolated populations of the San Bernardino kangaroo rat, such effects could have important conservation implications. Genetic studies of the San Bernardino are urgently needed.

Disease: The relationship of parasites and associates (e.g., viruses, bacteria, spirochetes, fungi, protozoa, etc.) in disease in *D. merriami* is not well understood, but various studies summarized by Whitaker *et al.* (1993) indicate that the species supports

and/or may be affected by a variety of organisms. While many of these “parasites” may be benign, others may cause disease and mortality that could have severe impacts on small, insular populations. Because of the enormous number of parasites and associates *D. merriami*, on a brief summary of the general types and number of genera and species are reported here. The reader is directed to Whitaker *et al.* (1993) for a more detailed description.

D. merriami is known to carry at least two fungi species, eight species of protozoa, four species of tapeworm (cestodes), 10 species of roundworm (nematodes), 10 species of mites, 34 species of chiggers, two species hard ticks, two species of sucking lice, one moth, and 22 species of fleas. The effects of these parasites and their associates on the health of *D. merriami* generally are unknown. Many may be benign, but some may be pathogenic and have deleterious effects on populations (Whitaker *et al.* 1993). Such effects in small, isolated populations would be particularly serious. The relationships between host and parasites, such whether they cause harm to the host, the geographic range of the parasites, and whether the number of parasites an individual carries is related to health, are all topics that require further study (Whitaker *et al.* 1993).

3.2 Other Sensitive Species

3.2.1 Parry’s Spineflower

SPECIES DESCRIPTION

Status

Parry’s spineflower is not state- or federally-listed. The species is on the California Native Plant Society’s List 1B with an R-E-D code of ?-2-3.

Habitat and Habitat Associations

Parry’s spineflower occurs within the alluvial chaparral and scrub of the San Gabriel, San Bernardino and San Jacinto Mountains, at elevations of 100 to 1,300 m above msl (Reveal and Hardham 1989).

Biology

Genetics: Parry’s spineflower comprises the Parryanae subsection of *Chorizanthe* in Polygonaceae (Reveal and Hardham 1989).

Reproduction: Parry’s spineflower has white flowers and blooms from April through June. The brown achenes are 2.5 to 3 mm long (Reveal and Hardham 1989).

Dispersal: No literature was available regarding dispersal mechanisms.

Demography: This prostrate to spreading plant is an annual species (Reveal and Hardham 1989).

Known Distribution

This species is known from the flats and foothills of the San Gabriel, San Bernardino and San Jacinto Mountains within Los Angeles, San Bernardino and Riverside Counties of southern California (Reveal and Hardham 1989). Parry's spineflower is possibly extirpated from Los Angeles County (CNPS 2001). Boyd (1999) notes that this species occurs in the Liebre Mountains, Los Angeles County.

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum, the UCR herbarium and available literature. There are four occurrences of Parry's spineflower recorded in the Wash Planning Area, two from the UCR herbarium and two from the Robertson's Ready Mix Project reports (Lilburn Corporation 1996). The species occurs along the floodplain of the Santa Ana River. The occurrences were mapped in chamise chaparral, chamise chaparral/non-native grassland, mature Riversidean alluvial fan sage scrub and intermediate/mature Riversidean alluvial fan sage scrub.

Special Biological Considerations

Parry's spineflower may be confused with other species of spineflowers, particularly *Chorizanthe procumbens* (CNPS 2001).

Threats to Species

Parry's spineflower is threatened by habitat loss as a result of urbanization (Reveal and Hardham 1989; CNPS 2001), mining and flood control practices (U.S. Fish and Wildlife Service, unpublished data).

3.2.2 Plummer's Mariposa Lily

SPECIES DESCRIPTION

Status

Plummer's mariposa lily is not state- or federally-listed. The species is on the California Native Plant Society's List 1B with an R-E-D code of 2-2-3.

Habitat and Habitat Associations

This species occurs on rocky and sandy sites, typically of alluvial or granitic material, in coastal scrub, chaparral, cismontane woodland, lower montane coniferous forest and valley and foothill grasslands at elevations from 90 m to 1,610 m (CNDDDB 2000; CNPS 2001).

Biology

Genetics: Plummer's mariposa lily is a member of subsection *Weediani*, section *Cyclobothra*, genus *Calochortus* and belongs to the Liliaceae (Ness 1989). Plummer's mariposa lily hybridizes with intermediate mariposa lily (*C. weedii* var. *intermedius*), also a member of subsection *Weediani*, where the two are sympatric in the San Jose Hills and Puente Hills (Ness 1989; CNPS 2001).

Reproduction: This species flowers from May through July (CNPS 2001). The inflorescence consists of two to six bell-shaped flowers. The pale pink or rose petals have a wide central band of long yellow hairs and are bearded on the inner face with long yellow hairs. Each petal also has a round gland; the gland is either glabrous or bordered with a ring of dense orange hairs. The erect capsules are 4 to 8 cm long (Munz 1974; Fiedler and Ness 1993). Information regarding pollinators of this species was not reviewed.

Dispersal: Information regarding dispersal of this species was not reviewed.

Demography: This perennial bulb has a fibrous coat (Ness 1989; Fielder and Ness 1993). The leaves are basal and vary in length from 20 to 40 cm and the stems are 30 to 90 cm high (Fiedler and Ness 1993). Information regarding the life span of this species was not reviewed.

Known Distribution

Plummer's mariposa lily is known from Ventura County, Los Angeles County, San Bernardino County and Riverside County (CNPS 2001).

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum, the UCR herbarium and available literature. There are 24 occurrences of Plummer's mariposa lily recorded in the Wash Planning Area; one occurrence was from the CNDDDB, two occurrences from the UCR herbarium, 20 occurrences were from the Robertson's Ready Mix Project reports (Lilburn Corporation 1996) and one occurrence were from the Sunwest Materials Project reports (Lilburn Corporation 1997). The species occurs along the floodplain of the Santa Ana River and Plunge Creek. Of the 22 occurrences, three were mapped in disturbed areas, one was mapped in chamise chaparral and one was mapped in the recharge basins. The remaining occurrences were mapped in immature and mature Riversidean alluvial fan sage scrub.

Special Biological Considerations

A bulb-bearing perennial, this species may not flower in very dry years and may be difficult to locate during surveys conducted in such a year. Flowering may also be suppressed by heavy infestations of weedy grasses.

Threats to Species

This species is threatened by urban development (CNPS 2001). Like other bulb-bearing perennials, this species is probably susceptible to damage from ground disturbance activities (e.g., discing).

3.2.3 Robinson's Pepper-Grass

SPECIES DESCRIPTION

Status

Robinson's pepper-grass is not state- or federally-listed. It is on the California Native Plant Society's List 1B with an R-E-D code of 2-3-2.

Habitat and Habitat Associations

Robinson's pepper-grass generally occurs in coastal sage scrub and chaparral habitats below 2850 feet (855 meters) AMSL. It is more typically observed in dry, exposed areas rather than beneath the shrub canopy or along creeks (Reiser 1994). Robinson's pepper-grass is also reported growing in non-native grassland and coastal sage scrub mixed with non-native grassland. Soil types reported as providing habitat include decomposed granite, gravelly, coarse sandy, sandy loam, and gabbroic clay (CNDDDB 2003).

Biology

Genetics: Robinson's pepper-grass is one of five subspecies of the annual pepper-grass (*Lepidium virginicum*) that occur in California. Variety *robinsonii* may be distinguished from the more common varieties of this species by its densely hairy stem and inflorescence and by having lobed leaves along the stem (Hickman 1993). Pepper-grass has 16 pairs of chromosomes, which is double the number of some congeners and equal to others (Smith 1938).

Reproduction: Robinson's pepper-grass flowers from January to April (Munz 1974). Pepper-grass (*L. virginicum*) seed viability of between 25 and 50 years has been demonstrated under laboratory conditions (Beal, 1905; Darlington, 1931).

Dispersal: No information on dispersal of Robinson's pepper-grass is available in the literature.

Demography: No information on demography of Robinson's pepper-grass is available in the literature.

Known Distribution

Robinson's pepper-grass is distributed from Santa Barbara County, California south to Baja California, Mexico at elevations below 2850 feet (855 meters). It generally occurs well inland from the coast, but is reported from Point Loma in San Diego and Santa Cruz Island (CNPS 2004).

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum, the UC Berkeley herbarium, the UCR herbarium and available literature. There is one 1987 report of Robinson's pepper-grass from the northeastern portion of the Wash Planning Area, north of the Santa Ana Wash and south of Greenspot Road; the population was reported in sage scrub on coarse, sandy soils. Although this was filed as an undetermined taxon in the UC herbarium (CNDDDB 2003), it is currently listed as Robinson's pepper-grass (UC Berkeley Herbarium 2004). Historically, Robinson's pepper-grass was reported from dry hillsides in the vicinity of San Bernardino by Parrish in 1884 and 1889 (CNDDDB 2003). No collections of Robinson's pepper-grass from the Wash Planning Area are present in the UCR herbarium (2004).

Special Biological Considerations

Robinson's pepper-grass is relatively small (up to eight inches tall) and is likely to require openings or sparse vegetation.

Threats To Species

Although CNPS regards Robinson's pepper-grass as seriously endangered in California, only erosion and feral herbivores on Santa Cruz Island are specifically listed as threats (CNPS 2004). Reiser considers the species likely to be more common than indicated by the number of collections because its' chamise chaparral habitat has not been extensively searched by botanists; he presumes that Robinson's pepper-grass is stable in southern California (Reiser 1994). Pepper-grass (*Lepidium virginicum*) is a weedy species with a cosmopolitan distribution, and Robinson's pepper-grass has been reported from sites that have been largely converted from coastal sage scrub to annual grassland (CNDDDB 2003). The primary threat to the species in southern California is likely to be direct loss of habitat through development, with relatively less threat from related impacts such as habitat fragmentation and competition from exotic species.

3.2.4 San Diego Horned Lizard

SPECIES DESCRIPTION

Status

San Diego horned lizard is a California Species of Concern. It is not state- or federally-listed. CDFG has reclassified this taxon as the *blainvillei* population, rather than as a separate subspecies, of *Phrynosoma coronatum*, and has adopted the common name "coast horned lizard" for the entire species. In this discussion, the *blainvillei* population is referred to as the San Diego horned lizard, and the species in general is referred to as coast horned lizard.

Habitat and Habitat Associations

San Diego horned lizard is found in a wide variety of habitats including coastal sage scrub, annual grassland, chaparral, oak woodland, riparian woodland, and coniferous forest. In the southern California coastal plain it had been most abundant in riparian and coastal sage habitats on old alluvial fans. In foothill and mountain areas with dense bush cover, it is restricted to pockets inland pockets of open microhabitat created by disturbance such as floods, fire, roads, grazed areas, or fire breaks (Jennings and Hayes 1994). Recorded observations of San Diego horned lizard in San Bernardino and Riverside Counties cite various forms of chaparral, Riversidean alluvial sage scrub, Riversidean sage scrub, sparse sycamore riparian woodland, juniper scrub, oak

woodland, grassland, remnant grape vineyards, and disturbed vegetation as habitat (CNDDDB 2003).

Biology

Genetics: Adaptation of horned lizards (*Phrynosoma* spp.) for myrmecophagy (ant-eating) appears to have resulted in the reduction or loss of several specialized tissues (the epipterygoid, coronoid process, an area posterior to the coronoid process, and the mandibular ramus), and a concomitant increase in the length of the tooth row. Behavioral information corroborates this specialization for myrmecophagy among horned lizards (Montanucci 1989).

Diet and Foraging: The San Diego horned lizard diet is dominated by native harvester ants (*Pogonomyrmex* spp.) (Pianka and Parker, 1975). It does not appear to eat non-native Argentine ants (*Linepithema humile*) that have replaced native ants in much of southern California (Jennings and Hayes, 1994; Ward, 1987). Other slow moving insects, such as beetles, flies, and caterpillars are consumed opportunistically (Presch 1969; Pianka and Parker 1975).

Individual coast horned lizards were found to eat from approximately 30 to over 100 harvester ants per day one with a maximum of 72 eaten at a single stop; with up to four different species of harvester ants consumed.. The coast horned lizards fed most often on ants that were not associated with nest discs or foraging columns and took only a few ants at any one place. Hatchlings fed on an average of three harvester ants per bout, with 20 to 30 minute pauses between feeding (Whitford and Bryant 1979).

Daily/Seasonal Activity: San Diego horned lizard was observed to emerge from burial sites in the substrate just prior to sunrise, as surface temperatures exceed 19°C, and to bask in the first rays of the sun (Heath, 1962, 1965; Hagar, 1992). The lizards may move upward in the sand until just their heads are exposed emerge completely and begin basking (Heath 1962). Whitford and Bryant (1979), however, did not observe activity until approximately two hours after sunrise, with most feeding and other activity confined to the morning hours. Horned lizards paused between feeding bouts for periods of 30 seconds to several minutes. Their feeding corresponded with the peak activity patterns of harvester ants, between the hours of 9 a.m. and 11 a.m. (Whitford and Ettershank, 1975; Whitford, *et. al.*, 1976). Through the middle part of the day, San Diego horned lizards positioned themselves in a shrub canopy where the ambient temps ranged from 35 to 40 degrees C (Whitford and Bryant, 1979) or bury themselves in the substrate, reemerging in the later afternoon to resume feeding, territorial, and reproductive activities (Heath 1965).

Olfactory cues may be important in San Diego horned lizard's activities such as courtship, feeding, sex recognition, and conspecific interactions; they appear to mark sites by partially extruding the cloaca and rubbing it back and forth on the substrate (Tollestrup 1981).

Reproduction and Development: In southern California, the male reproductive cycle begins during mid to late March and ends in June as testes decrease in size (Goldberg 1983). Female San Diego horned lizards typically lay a single clutch of 6 - 17 (most commonly 11 to 12.5) eggs between May and July each year (Stebbins 1954; Howard 1974; Goldberg 1983). Hatchlings appear in late July to early August, and require two to three years to reach reproductive age (Stebbins, 1954; Howard, 1974; Pianka and Parker, 1975; Goldberg, 1983). San Diego horned lizard has the potential to produce multiple clutches (1983).

Survival: The San Diego horned lizard's most common defense is to lie motionless, depending on their cryptic appearance (Jennings and Hayes 1994). Klauber (1939) documented change in body coloration to match the soil or sand on which they were found. Other defensive methods include hissing, inflating lungs to increase apparent size (Pianka and Parker 1975; Munger 1984; Sherbrooke 1981), raising their horns by lowering their snout (Pianka and Parker 1975; Sherbrooke 1981), squirting blood from the corner of the eye (which seems to repel dogs and cats) (Presch 1969; Pianka and Parker 1975), tilting the body when irritated (Milne and Milne 1950; Smith 1946; Tollestrup 1981), presenting a bristling of scales of the back while standing well up on the legs (Bryant 1911), and running a short distance before flattening out or burrowing several centimeters under the ground (Presch 1969). When the San Diego horned lizards flattens its body, it usually tucks its head down, exposing its horns, and often charges the enemy (Winton 1916). Learned avoidance of horned lizards by predators is suggested by reports of snakes dying while trying to swallow horned lizards (Klauber 1972; Milne and Milne 1950; Van Denburgh 1922; Vorhies 1948; Wright and Wright 1957).

Dispersal: No information on dispersal of San Diego horned lizard is available.

Socio-Spatial Behavior: San Diego horned lizards use several displays for species recognition, courtship, and sex; including head-bobs, push-ups, curling up the tail, and scratching. Displays between males are usually performed from an elevated perch such as a gopher mound or cow dung, and are characterized by a frequency increase in head-bobs and push-ups, and by the use of the rocking display. One male would then run toward the other, each continuing to display, subordinate males curl up the tail and move out of the area. No biting or combat with horns was observed (Tollestrup 1981).

Horned lizards have limited home ranges, occupying smaller areas than they would if they moved randomly. Home range overlap is reduced and contrary to expectation, overlap between sexes tended to be less than overlap between individuals of the same sex (Munger, 1984). The limited home range overlap may be due to a low-level home range defense in the form of head bobbing; or mutual voluntary avoidance may be practiced because the areas occupied by other horned lizards are likely to have been recently harvested (Munger 1984).

Coast horned lizards moved an average of 46.8 meters per day (range = 9 - 91 meters), moving over a zigzag course during a day but rarely crossed its own trail (Whitford and Bryant, 1979)

Community Relationships: Horned lizard (*P. cornutum*) foraging allows for maximization of prey availability over a period of weeks or a month rather than hours or a day. Horned lizards appear to be limited by the availability of harvester ants, using the harvester ants at or close to the maximum exploitation level, making them unavailable to other potential predators (Whitford and Bryant, 1979). In over six years of studying these ants, Whitford and Bryant have reported only two incidents of harvester ant predation by other species: one by a robber fly (Asilidae) and one by a sun spider (Solpugidae). The San Diego horned lizard is presumed to fill the same habitat niche in southern California as *P. cornutum* does in Texas.

Harvester ant foraging in the presence of horned lizard varied from no response by *P. desertorum*, to an avoidance response (twice observed) by a column of *P. rugosus* foraging, involving ants in the column becoming immobile, assuming a vertical position on the soil surface or while clinging to a grass blade, for 10 to 15 minutes (Whitford and Bryant, 1979).

Removal of 50% or more of the foraging harvester ants (*P. rugosus*) during daytime caused colonies to cease activity for up to ten days; no effect was found at lower levels of simulated predation during the daytime or from any level of removal at night. Simulated predation also slowed the rate of harvester ant foraging. (Whitford and Bryant, 1979).

Known Distribution

Historically, San Diego horned lizard was distributed from the Transverse Ranges in Kern, Los Angeles, Santa Barbara, and Ventura counties southward through the Peninsular Ranges of southern California to Baja California (Jennings, 1988). San Diego horned lizard has apparently disappeared from about 45% of its former range in southern California, in particular on the coastal plain where it was once common (Hayes and Guyer, 1981) and in riparian and coastal sage scrub habitats on the old alluvial fans

of the southern California coastal plain (Bryant, 1911, Van Denburgh, 1922). San Diego horned lizard now ranges from the Transverse Ranges south to the Mexican border and west of the deserts, occurring at scattered sites along the extreme western desert slope of the Peninsular Ranges (Jennings, 1988). The known elevation range of this species is from 30 feet (10 m) at the El Segundo dunes (Los Angeles County) to approximately 7,100 feet (2,130 m) at Tahquitz Meadow, on San Jacinto Mountain, in Riverside County.

The San Diego horned lizard (i.e. *blainvillei* population) is thought to intergrade with the *frontale* population in extreme southern Kern county and northern Santa Barbara, Ventura, and Los Angeles counties (Reeve, 1952; Montanucci, 1968; Jennings, 1988).

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum, the UCR herbarium and available literature. According to the CNDDDB (2004), two records of San Diego horned lizard were reported within the Wash Planning Area from a 1991 study by S. Hager. Ten individuals, including four adult males and four adult females, were observed in intermediate Riversidean alluvial fan sage scrub in the Santa Ana River wash near the western end of the Wash Planning Area. A second group of four adults, one juvenile, and ten hatchlings was observed in an area mapped as mature Riversidean alluvial fan sage scrub. The San Diego County Natural History Museum database contains eight occurrences, observed between 1997 and 2000. These occurrences were located in intermediate and mature Riversidean alluvial fan sage scrub.

Special Biological Considerations

Argentine ants are a highly aggressive species that out-competes native harvester ants, the primary prey item of the San Diego horned lizard. Argentine ants have flourished in southern California where supplemental water sources are present. The availability of San Diego horned lizard prey items is likely to be inversely correlated with suitable Argentine ant habitat, particularly artificially moist sites, within and adjacent to the Wash Planning Area.

Threats to Species

The specialized diet and habitat requirements, site fidelity, and cryptic defense behavior make San Diego horned lizard highly vulnerable. Commercial collecting and habitat loss due to agriculture and urbanization are the main reasons cited for the decline of this taxa. Most surviving populations inhabit upland sites with limited optimal habitat. Many of these sites are on marginally suitable Forest Service land (Jennings and

Hayes, 1994). The greatest threat to San Diego horned lizard is the loss of its predominant food source, the harvester ant, through competition from the exotic Argentine ant. Argentine ants colonize around disturbed soils associated with building foundations, roads and landfills, and expand into adjacent areas, eliminating native ant colonies (Ward, 1987), causing loss and fragmentation of San Diego horned lizard foraging habitat. Fire, grazing, off-road vehicles, domestic cats, and development are other stressors (Jennings and Hayes, 1994). This taxon is unable to survive habitats altered by development, agriculture, off-road vehicle use, or flood control structures (Goldberg, 1983).

3.2.5 Western Spadefoot

SPECIES DESCRIPTION

Status

The western spadefoot is a California Species of Concern and is considered sensitive by the Bureau of Land Management. This species is not state- or federally-listed.

Habitat and Habitat Associations

Western spadefoot may be found in coastal sage scrub, chaparral, and grasslands habitats, but is most common in grasslands with vernal pools or mixed grassland/coastal sage scrub areas (Holland and Goodman 1998). Within these habitats, western spadefoot requires rain pools with water temperatures between 9°C - 30°C in which to reproduce (Brown 1966, 1967), and that persist with more than three weeks of standing water (Feaver 1971) in which to metamorphose successfully. Additionally, Holland and Goodman (1998) report that riparian habitats with suitable water resources may also be utilized. Rain pools must lack fish, bullfrogs, and crayfish in order for western spadefoot to successfully reproduce and metamorphose (Jennings and Hayes 1994). Though not observed specifically for this taxon, soil characteristics of burrow refuge sites likely become fairly hard and compact during the period of summer estivation (Jennings and Hayes 1994, Ruibal *et al.* 1969). *S. hammondi* estivates in upland habitats adjacent to potential breeding sites in burrows approximating 1 meter in depth (Stebbins 1972).

Biology

Genetics: Genetic variation across the range of western spadefoot has not been studied (Jennings and Hayes 1994). Differentiation of western spadefoot from *S. multiplicatus* occurred in 1976 and was based on morphological, vocalization, and reproductive differences (Brown 1976).

Diet and Foraging: Western spadefoot tadpoles consume planktonic organisms and algae, but are also carnivorous and will forage on dead vertebrates and invertebrates (Bragg 1964). Also, spadefoot tadpoles are known to pursue and eat fairy shrimp (Bragg 1962). The capability of tadpole cannibalism in the genus *Scaphiopus*, is one of many adaptations that allows for breeding in temporary pools (Low 1976). Spadefoot toads are more likely to express a carnivorous/ cannibalistic phenotype when reared with multiple broods that include non-siblings/kin. When raised exclusively with kin, the carnivorous phenotype can be suppressed. Studies on *S. bombifrons* and *S. multiplicata* showed some differentiation in the trigger for carnivorous behavior. *S. multiplicata* individuals were more likely to express the carnivore phenotype in mixed sibship groups than in pure sibship groups. While *S. bombifrons* tadpoles were significantly more likely to express the carnivore phenotype when reared alone than in pure sibship groups (Pfennig and Frankino 1997). Regardless, both species exhibited this phenotype independent of food availability or sibship differences in size or growth rate, and waterborne chemical signals were enough to initiate expression of the carnivore phenotype (Pfennig and Frankino 1997). Farrar and Hey (1997) found that carnivorous spadefoot toads developed longer snouts, larger beaks with modified cusps, shorter intestines with fewer loops than omnivores and they feed on fairy shrimp. Studies conducted on *S. couchii* show that a constant high rate of food availability allows for the largest, and presumably, the most fit metamorphs (Newman 1994).

Adult spadefoot toads in general, are known to consume butterfly and moth larvae, beetles, termites, and ants (Dimmitt and Ruibal 1980, Whitaker *et al.* 1977). Additional food items include crickets, flies, ants, earthworms and other invertebrates (Stebbins 1972, Morer and Gullin 1992). Anderson *et al.* (1999) found that two different spadefoot taxa consumed between 12 and 20 different invertebrate taxa during a two years study. Western spadefoot is able to consume approximately 11% of their body mass at a single foraging event (Dimmitt and Ruibal 1980). A study of the tongue musculature associated with foraging in adult *S. multiplicata* was completed by O'Reilly and Nishikawa (1995) as part of evolutionary biology research.

Daily/Seasonal Activity: Western spadefoot is almost entirely nocturnal (Holland and Goodman 1998), with most above ground movement and breeding occurring during rainy nights (Ziener *et al.* 1988). Typically, spadefoot toads are not found above the surface, instead they are found in underground burrows (Stebbins 1972) for most of the year. Spadefoot toads remain underground 8 to 10 months of the year (Jennings and Hayes 1994, Holland and Goodman 1998, Storey *et al.* 1999) following which adults emerge from underground burrows during relatively warm (≥ 10.0 - 12.8 degrees Celsius) rainfall events to breed, typically from January through March; however, they may also emerge in any month between October and April if rain thresholds are met

(Stebbins 1972, Morey and Guinn 1992, Jennings and Hayes 1994, Holland and Goodman 1998).

A few studies focus on the mechanisms of spadefoot toad estivation. Storey *et al.* (1999) reviewed genes that were induced or upregulated in two-month estivating female spadefoot toads during estivation. The focal protein, riboflavin binding protein, is produced by the liver in birds, mammals, and reptiles in order to bind plasma riboflavin and load the vitamin into eggs or fetus. The liver-specific protein allows the toad to cache vitamin production over the estivation period in preparation of the breeding event after emerging. Transitional mechanisms between dormant and active states in the toad, and resulting metabolism shifts, appear to rely on the reversible phosphorylation control of intermediary metabolism enzymes (Cowan and Storey 1999) for protection of estivating muscle tissue. To further protect the estivating body, enzymatic and metabolite antioxidant defenses may be modulated in accordance with estivation status (Grundy and Storey 1998). Finally, Grundy and Storey (1994) studied the effects of stored urea on estivating spadefoot toad, and found that the high concentrations of urea minimized dessication as a result of increased salt concentrations.

Reproduction and Development: Spadefoot tadpoles exhibit numerous adaptations for breeding in temporary pools: rapid embryonic and larval development, tadpole cannibalism, production of growth inhibitors by tadpoles, and high heat tolerance of tadpoles (Loe 1976). After periods of warm rains, spadefoot toads emerge from burrows and form explosive, and sometimes large (>1000 individuals; Jennings and Hayes 1994) aggregations. This typically occurs in late-winter and early-spring, but may also occur during the fall (Storer 1925, Feaver 1971, Jennings and Hayes 1994). Caching of riboflavin during the nine- to 10-month estivation period, allows spadefoot toads to maintain an endogenous vitamin pool which may be linked with maturation of eggs in preparation for the explosive breeding period after emergence from estivation (Storey *et al.* 1999). Zeiner *et al.* (1988) indicates that artificial irrigation may elicit advertisement (reproductive) vocalizations during any month. Holland and Goodman (1998) note that breeding efforts are probably tied to the amount of rainfall. Sullivan and Fernandez (1999) found that breeding activity of *S. couchii* was restricted to significant rainfall events (>25 mm of rainfall within 24 hours). Because the critical thermal minimum is ninerees Celsius (Brown 1966), spadefoot toads wait until water temperature is at least 10 degrees Celsius before egg deposition (Jennings and Hayes 1994). Eggs are deposited in irregular small cluster, about 25-30 centimeters in diameter (Holland and Goodman 1998), attached to vegetation or debris (Storer 1925) in shallow temporary pools or sometimes ephemeral streamcourses (Stebbins 1985, Jennings and Hayes 1994). Egg clusters rarely number above 42 (Jennings and Hayes 1994). The rate of egg hatching is water temperature dependant (Brown 1967), however, eggs are usually hatched within six days. Complete development can rapidly

occur within three weeks (Holland and Goodman 1998), but may last up to 11 weeks (Burgess 1950, Feaver 1971, Jennings and Hayes 1994).

The rate of development is regulated by water temperature, water evaporation, and food resources (Holland and Goodman 1994, Denver 1998, Denver *et al.* 1998, Newman 1998). Tadpoles subjected to water volume reduction showed significant acceleration of metamorphosis (Denver *et al.* 1998), but the rate of accelerated development was determined by rate of water reduction and was reversible (decelerated development) by replacement of water. Thermal differences, compound concentration, and chemical or physical interactions to conspecifics did not play a role in the rate of acceleration. An accelerated metamorphosis appears to be a response to reduce swimming volume and proximity to water surface (Denver *et al.* 1998).

Newman (1998) studied the effects of temperature and food on the development of *S. couchii*. He found that the age at metamorphosis was primarily determined by the early food regime and size at metamorphosis was determined by food level late in the larval period, but response due to food availability was dependant on environmental factors. The interaction between food availability and temperature gleaned the following results: **(1)** at high temperature, high initial food availability, and low tadpole density, development was rapid and tadpoles switched from high to low food concentrations metamorphosed at about the same time and size as those at a constant food concentration; **(2)** under high temperatures, high initial food concentrations, and at high tadpole density, tadpoles switched to low food concentrations metamorphosed somewhat earlier and smaller than tadpoles kept at high food concentrations; **(3)** at low temperature and low tadpole density, tadpoles metamorphosed much smaller and earlier; **(4)** at low temperature and high tadpole density, tadpoles metamorphosed smaller and later; and **(5)** the combination of high tadpole density and constant low food availability prevented metamorphosis at high temperature and few metamorphs at low temperature.

The metamorphic response in spadefoot toads to pond dessication is initiated by the activation of the thyroid and interrenal axis (Denver 1997), and the response is rapid (within 48 hours) (Denver 1998). Seasonal expression of secondary sex characteristics in *S. couchii* is associated with plasma elevations in androgens (Harvey and Propper 1997).

Survival: No data is available for western spadefoot, however Sullivan and Fernandez's (1999) breeding study of four desert amphibians (including *S. couchii*) between 1990 and 1995, found that all exhibited rapid growth to maturity but they were not long-lived.

Dispersal: No data are available on the movement ecology or colonization abilities of western spadefoot (Jennings and Hayes 1994). However, Zeiner *et al* (1988) states

that after transforming in the late spring, juvenile toads disperse after a short period of time.

Socio-Spatial Behavior: Though little is known of the socio-spatial behavior of western spadefoots, they likely do not move far from their breeding pool during the year (Zeiner *et al.* 1988), and it is likely that their entire post-metamorphic home range is situated around a few pools. Western spadefoot may be aggressive at breeding sites (Whitford 1967) which is likely due to territorial defense of a small breeding zone during the explosive breeding season. Tadpoles may compete for food resources or space with other amphibian larvae such as western toad and Pacific treefrog, however once metamorphosed they are likely to escape predators and competitors (Zeiner *et al.* 1988).

Community Relationships: There is no information regarding community relationship and western toads, however, *S. bombifrons* may be cannibalistic when growing with non-siblings (Pfennig *et al.* 1993; Pfennig 1999) and other spadefoot toads are known to hybridize with *S. couchi* (Wasserman 1964; Blair 1947). Since these species appear to be quite similar to western spadefoot in habits and ecology, it is possible that these characteristics are also shared.

Known Distribution

Western spadefoot is a California near endemic ranging from Shasta County southward into Baja California (Stebbins 1985). Its known elevation range extends from near sea level to 1,500 m (Zeiner *et al.* 1988, Ervin *et al.* 2001). The known range of western spadefoot is restricted to west of the Sierran-desert range axis (Myers 1944). About 80% of the habitat once known to be occupied by western spadefoot in southern California has been developed or converted to uses incompatible with successful reproduction or recruitment (Jennings and Hayes 1994).

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum and available literature. The San Bernardino County Natural History Museum has two known occurrences of western spadefoot recorded in the Wash Planning Area between 1997 and 2000. These occurrences were recorded in disturbed habitat and intermediate Riversidean alluvial fan sage scrub near the existing SBVWCD recharge basins.

Special Biological Considerations

Western spadefoot enters the water only to breed (Dimmitt and Ruibal 1980a). Western spadefoot emerges from burrows of at least one meter depth, following warm rains (10.0 - 12.8°C) in early spring and fall (Stebbins 1972). The taxon may, however, become surface active any time between October and April if enough rain has fallen (Morey and Guinn 1992). However, by late June and early July, toads have been observed to emerge after light rains, suggesting that by late June some toads may be active in their burrows and come to the surface at night (Ruibal *et al.* 1969). Spadefoots can absorb water through the skin from soil more effectively than any other amphibian (Ruibal *et al.* 1969).

Much study has been generated by Pfennig (1990, 1993) regarding the tendency for some tadpoles to become primarily carnivorous and even cannibalistic while most others remain omnivorous detritus eaters. Those tadpoles that leave the natal area are at most risk of cannibalism. In general, tadpoles are algae and detritus feeders, but they will occasionally eat fairy shrimp, mosquitoes, and smaller tadpoles. Adult western spadefoot will eat ants, flies, beetles, moths, snails, grasshoppers, spiders, and just about anything large enough to see and small enough to swallow (Whitaker *et al.* 1977). Some desert spadefoots are capable of consuming enough food in a single feeding to provide it with energy reserves for more than one year (Dimmitt and Ruibal 1980a), but western spadefoot probably requires several feeding events before it has gathered a year's fat reserves.

Threats to Species

The continued placement of mosquito fish by mosquito abatement programs in rain pools threatens some populations (Jennings and Hayes 1994). Bullfrogs emigrating into rain pool breeding sites may also pose a threat (Hayes and Warner 1985; Morey and Gullin 1992). By far the largest threat is continued conversion of habitat in southern California. Grazing, off-road vehicles, mining, and projects which impact fluvial processes in burrow areas have a significant impact on local populations. Emergence from dormancy depends on low frequency sound caused by rainfall events, but work completed by Dimmitt and Ruibal (1980b) showed that the vibration caused by an electric motor consistently induced 100% emergence from dormancy under very arid conditions.

3.2.6 Burrowing Owl

SPECIES DESCRIPTION

Status

The burrowing owl is a California Species of Concern and is designated by the U.S. Fish and Wildlife Service as a Bird of Conservation Concern and by the Bureau of Land Management as a sensitive species. This species is not state or federally-listed as threatened or endangered.

Habitat and Habitat Associations

The burrowing owl occurs in shortgrass prairies, grasslands, lowland scrub, agricultural lands (particularly rangelands), prairies, coastal dunes, desert floors, and some artificial, open areas as a year-long resident (Haug, *et al.* 1993). They may also use golf courses, cemeteries, road allowances within cities, airports, vacant lots in residential areas and university campuses, fairgrounds, abandoned buildings, and irrigation ditches (Haug, *et al.* 1993; Hayworth 1990 pers. obs.). They may also occur in forb and open shrub stages of pinyon-juniper and ponderosa pine habitats (Zeiner, *et al.* 1990). They require large open expanses of sparsely vegetated areas on gently rolling or level terrain with an abundance of active small mammal burrows. As a critical habitat feature need, they require the use of rodent or other burrows for roosting and nesting cover. They may also dig their own burrow in soft, friable soil (as found in Florida) and may also use pipes, culverts, and nest boxes where burrows are scarce (Robertson 1929). The mammal burrows are modified and enlarged. One burrow is typically selected for use as the nest, however, satellite burrows are usually found within the immediate vicinity of the nest burrow within the defended territory of the owl.

Biology

Genetics: The burrowing owl has been variously placed in the monotypic genus *Speotyto* or in *Athene*, where it has three congeners (Haug, *et al.* 1993). Comparison with other karyotypes in the literature suggests that the burrowing owl should be in a separate genus, *Speotyto*, as has been done for a number of years although it is frequently still referred to as *Athene* (Schmutz and Moker 1991).

Diet and Foraging: The burrowing owl is a crepuscular hunter with a prey base including invertebrates and small vertebrates (Thomsen 1971). They may hunt by using short flights, running along the ground, hovering or by using an elevated perch from where prey is spotted. They typically forage in short-grass, mowed, or overgrazed pasture, golf courses and airports (Thomsen 1971).

They are a relatively opportunistic forager (Haug, *et al.* 1993). Their diet is composed of a variety of foods included *Peromyscus*, *Microtus* and beetles. Beetles occur within their diet with more frequency; however, based on biomass, *Peromyscus* is dominant with *Microtus* appearing second in overall biomass (Marti 1974). Although they eat mostly insects and small mammals, they also may take reptiles, birds, and carrion.

During the breeding season, there are significant declines in the percentage of vertebrate prey in the diet and increases in the invertebrate prey (Haug, *et al.* 1993).

Daily Activity: The burrowing owl is primarily a diurnal species with crepuscular hunting habits (Thomsen 1971). They may move the location of their perch in order to thermoregulate by perching in open sunlight in early morning and then moving to shade or to the burrow, when temperatures are hot (Coulombe 1971).

Reproduction: The burrowing owl usually nests in an old burrow of a ground squirrel, or other small mammal, and may also use the burrow of badgers and marmots. It may dig its own burrow in soft soil. The nest chamber is lined with excrement, pellets, debris, grass, feathers; sometimes it is unlined. Pipes, culverts, and nest boxes are used where burrows are scarce (Robertson 1929). The male gives a courtship display and notes in front of the burrow. Breeding occurs from March through August, with a peak in April and May. The clutch size is 6-11 eggs, with an average of 7-9 eggs; this clutch size may increase to the north (Bent 1938). The young emerge from the burrow at about two weeks, and they fly by about four weeks (Zarn 1974). Martin (1973) reported 95% of the young fledged, and a mean reproductive success of 4.9 young per pair. The species is semi-colonial; it is probably the most gregarious owl in North America.

Nest success was 50 to 57% at a site in Oregon with desertion being the major cause of nest failures and typically was related to the proximity to other nesting pairs. Burrow sites with good horizontal visibility and little grass coverage were preferred. Elevated perches were used in habitat with average vegetation height greater than 5 centimeters and not in habitats with vegetation less than 5 centimeters. The elevated perches presumably improved the burrowing owl's ability to detect both predators and prey by increasing their horizontal visibility (Green and Anthony 1989). MacCracken *et al.* (1985) found that nest burrows were in soils with a greater sand content than non-nest burrows, suggesting that selection for soil type may occur. All nest burrows found to be reused in a study in Oregon were in silty loam (Green 1983).

Survival: The minimum annual survival rates in Florida average 68% for adult males, 59% for adult females and 19% for one year old owls (Millsap and Bear 1992). In southern California, the apparent survival rates are 30% for juveniles and 81% for adults (Thomsen 1971). One banded bird survived to 8 years 8 months (Kennard 1975). Collisions with autos may be a significant cause of mortality (Remsen 1978).

Dispersal: A total of 92% of 555 owls that were banded at a nesting area were never re-encountered after the year in which they were banded. The 8% that returned to the natal area after being banded, returned one or more years after banding and stayed in the natal area for 2 to 4 breeding seasons (Lutz and Plumptre 1999). Returns of one year old owls were located 2.4 to 26.4 kilometers from the natal nest (Haug *et al.* 1993).

Socio-Spatial Behavior: The home range may vary from 0.1 to 4 acres (mean is 2 acres) with an average distance between burrows of 436 feet (Thomsen 1971, Martin 1973). Territory size is directly proportional to the available habitat and burrow availability (Haug *et al.* 1993).

Community Relationships: Predators include prairie falcons, red-tailed hawks, Swainson's hawks, ferruginous hawks, northern harriers, golden eagles, foxes, coyotes, and domestic dogs and cats (Martin 1973). Fleas, lice, and feather mites are common ectoparasites (Zeiner *et al.* 1990).

They require an abundance of active small mammal burrows. The availability of numerous small mammal burrows is a major factor in determining whether an area with apparently suitable habitat will support burrowing owls (Coulombe 1971). Burrowing owls rarely use areas unoccupied by colonies of burrowing mammals (Zarn 1974).

Potential competition with other owl species is avoided by the burrowing owls habit of hunting at a crepuscular time period and using other prey species including insects in their diets (Marti 1974).

Known Distribution

The burrowing owl breeds from southern interior British Columbia (nearly extirpated), southern Alberta, southern Saskatchewan (extirpated from a portion of the province), and southern Manitoba (extirpated from a portion of the province), south through eastern Washington, central Oregon, and California to Baja California, east to western Minnesota, northwestern Iowa, eastern Nebraska, central Kansas, Oklahoma, eastern Texas, and Louisiana, and south to central Mexico. The winter range is much the same as the breeding range, except that most burrowing owls apparently vacate the northern areas of the Great Plains and Great Basin (Haug, *et al.* 1993). The burrowing owl winters south regularly to El Salvador (*e.g.*, AOU 1998, 2003).

Historical changes in the distribution of the burrowing owl include the recent extirpation from British Columbia for which the last confirmed sighting was in 1979. Elsewhere in Canada and the north-central U.S., the range has contracted slightly southward, westward, and eastward (Haug *et al.* 1993). In Florida, the range has expanded northward, nearly to Georgia since the 1950s (Courser 1979).

Zeiner *et al.* (1990) describe the distribution, abundance, and seasonality of the burrowing owl within California as follows. It is a year-long resident formerly common in appropriate habitats throughout the state, excluding the humid northwest coastal forests and high mountains. It is present on the larger offshore islands and is found as high as

1,600 m (5,300 ft) in Lassen County. In California, burrowing owls are restricted to the central valley extending from Redding south to the Grapevine, east through the Mojave Desert and west to San Jose, the San Francisco Bay area, the outer coastal foothills area which extend from Monterey south to San Diego and the Sonoran desert (Grinnell and Miller 1944). It is a resident in the open areas of the lowlands over much of the southern California region (Garrett and Dunn 1981). It is greatly reduced in number within the lowlands of Riverside County and appears to be resident within the region although there is some movement of more northerly birds into the southern and coastal parts of the region (Garrett and Dunn 1981).

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum and available literature. The San Bernardino County Natural History Museum has two known occurrences of burrowing owl recorded in the Wash Planning Area between 1997 and 2000. These occurrences were recorded in disturbed habitat and intermediate Riversidean alluvial fan sage scrub along the Santa Ana River within the water conservation areas.

Special Biological Considerations

Physiological ecology studies have shown that the burrowing owl is able to dissipate 135% of their heat production by use of pulmocutaneous evaporation facilitated by gular flutter. This allows the species to use areas that may have air temperatures greater than their body temperature. They also were found to have different emissivities of their feathers depending on the season of year. During the winter, the emissivity of the plumage is greater thus allowing them to augment their metabolic heat production with solar radiative heat gain (Coulombe 1970).

The importance of retaining colonies must be stressed, as this species appears to have evolved as a colonial species in association with burrowing mammal communities (Dyer 1987). Minimum viable colony size is unknown. While these owls appear to adapt fairly well to human presence in some cases, *i.e.*, airport runways and other human modified open spaces, the continued presence of active mammal-created burrows is essential. In Oklahoma, the removal of prairie dogs allowed deterioration of burrows, making them unsuitable for nest burrows after one year (Butts 1973 as cited in Zeiner *et al.* 1990). Rodent eradication programs may reduce the consistent availability of high and moderate function habitat. The remaining habitat is often roadside drainage ditches, increasing potential for significant losses to vehicle collisions (Remsen 1978). The available soil type appears to be a factor in nest burrow selection (see Reproduction section above).

The burrowing owl was shown to choose moderately to heavily grazed grasslands for nesting and roosting and avoided cultivated fields. Where grassland patches were isolated in cultivation areas, the owls dispersed late, for shorter distances and less often. Mortality rate has been shown to be high in these systems. These changes from pasture to cultivation appear to be resulting in a decline of the species (Clayton and Schmutz 1999). It is also important to determine what type and where within the region owls are selecting burrows before the area is disturbed and before it is decided to provision them with artificial burrows. Burrowing owls produced fewer young when occupying a new burrow, and when using burrows in disturbed areas. They produced more young when using artificial burrows but produced fewer fledglings than natural burrows, thus the actual productivity decreased for the artificial burrows (Botelho and Arrowood 1998).

The role of food in limiting the number of offspring fledged from nests has been experimentally investigated in the burrowing owl (Wellicome 1997). Food-supplemented owls laid slightly larger clutches and produced eggs of higher volume but did not show higher hatching success or produce more hatchlings than did the unsupplemented birds. Therefore, although food intake may restrict the number of eggs that burrowing owls lay, the total number of young produced at a nest is constrained by food only during the nestling period. Food intake is thus more limiting during brood rearing than during egg laying (Wellicome 1997).

Urban sites can act as unintentional preserves and support owl populations if habitat features necessary for owls are provided. This is supported by the documented population at Moffett Field in Santa Clara County California. The population has established itself and is using nest burrows under cement or other hard surfaces. The adult density, number of young fledged or pairs with emergent young is not different at Moffett Field compared to other intentional preserve areas (Trulio 1997).

Human activities have had a beneficial effect in Florida where mowing, grazing of cattle and wetland drainage have increased the species' range. Residential and industrial areas currently support the largest concentrations of the species in Florida (Haug *et al.* 1993).

Because of the intense pressure for urban development within suitable burrowing owl nesting and foraging habitat in California, conflicts between owls and development projects often occur. Owl survival can be adversely affected by disturbance and foraging habitat loss even when impacts to individual birds and nest/burrows are avoided (CDFG 1995). The Staff Report on Burrowing Owl Mitigation (CDFG 1995) outlined the protocol for determining impact assessment. The project site and a 150-meter buffer should be surveyed according to the survey protocol and impacts to the owl should be considered to occur if there is disturbance within 50 meters of a burrow,

or there is destruction of natural or artificial burrows, or there is destruction of foraging habitat within 100 meters of a burrow. Mitigation measures should include the provision of 6.5 acres of foraging habitat per pair, provision of two burrows for each burrow impacted, relocation of owls (Trulio 1995), and avoidance of the nesting season.

Given the extraordinary, precipitous decline of this species in cismontane southern California (Grinnell and Miller 1944; Sexton and Hunt 1979; Garrett and Dunn 1981), it cannot be assumed that preferred habitat patches (*e.g.*, dry, level grasslands and open areas with suitable nesting substrates) within the will continue to accommodate the species in numbers similar to those in past years (Grinnell and Miller 1944). For instance, this species, fairly common in the Prado Basin and environs as recently as 1986, is now rare at that locale (Hays 1999 pers. obs.).

The following have been suggested as management strategies (Green 1983): protection of burrowing mammal populations; wood or plastic nest boxes and tunnels; artificial perches which provide hunting and predator observation sites; vegetation management through fire or grazing; and relocation of owls. Other management strategies include: reduce mortality on the breeding grounds, increase productivity, protect and manage the nesting habitat, monitor the populations, manage migration and wintering areas, conduct release programs, and develop public support (Hjertaas 1997).

Threats to Species

The threats to the burrowing owl include conversion of grassland to agriculture, other habitat destruction, predators, collisions with vehicles, and pesticides/poisoning of ground squirrels (Grinnell and Miller 1944, Zarn 1974 cited in Zeiner *et al.* 1990, Remsen 1978). A ranking by the resource agencies of the most important threats to the species included loss of habitat, reduced burrow availability due to rodent control, and pesticides (James and Espie 1997).

The burrowing owl was formerly common in appropriate habitats throughout the state, excluding the humid northwest coastal forests and high mountains. Population numbers have markedly reduced in recent decades (James and Ethier 1989; Zeiner *et al.* 1990). The primary threats to the species include the loss of natural habitat due to urban development and agriculture and the expressed effects of insecticides and rodenticides within occupied habitat. The use of insecticides may reduce the availability of their primary prey. Pesticides may have secondary adverse effects through contamination. The pesticide Carbofuran has been demonstrated to have negative impacts; Sevin is likely a safer pesticide (Hjertaas *et al.* 1995; Blus 1996). The loss of burrowing mammal colonies (due to rodenticides or other means) and the crushing of burrows by heavy equipment and ground maintenance machinery remain problematic. This species is usually associated with flat or shallow slopes on loamy soils; these areas are also

attractive to agriculture, as well as residential and industrial development. Shooting losses may be significant (Remsen 1978).

The burrowing owl received official status as Endangered in Canada as of 1986. Burrowing owls have gone from locally common to virtually extirpated in Minnesota in 50 years (Johnsgard 1988). The number of burrowing owl breeding pairs in central, western, and southern California have drastically declined in the last 50 years; during the 1980's the decline was probably greater than 70% (DeSante and Ruhlen 1995). The species appears to be seriously threatened with extirpation from central, western, and southern California because of the extent and intensity of development (DeSante and Ruhlen 1995).

3.2.7 Cactus Wren

SPECIES DESCRIPTION

Status

The cactus wren is a California Special Concern species and is listed federally by the U.S. Fish and Wildlife Service as a Bird of Conservation Concern and by the USDA Forest Service as a sensitive species. This species is not state or federally-listed as threatened or endangered.

Habitat and Habitat Associations

The cactus wren is an obligate, non-migratory resident of the coastal sage scrub plant community (as defined by Westman 1983 and O'Leary 1990). It frequents deserts and other arid terrain with thickets, patches, or tracts of larger, branching cacti, stiff-twigged, thorny shrubs, and small trees (Grinnell and Miller 1944). In other areas, it is considered an inhabitant of the Chihuahuan, Mojave, and Sonoran deserts and Tamaulpais thorn-shrub communities. It may also be considered a resident of scrubby flats, cactus and mesquite lowland areas, brushy mesas, gulches, hills, and canyons in Texas, desert riparian, creosote bush and large arroyos in Nevada (Proudfoot *et al.* 2000). It is closely associated with three species of cacti and occurs almost exclusively in thickets of cholla (*Opuntia prolifera*) and prickly pear (*Opuntia littoralis* and *Opuntia oricola*) dominated stands of coastal sage scrub below 457 meters in elevation on mesas and lower slopes of the coastal ranges (Proudfoot *et al.* 2000). Although it lives over a wide range from Texas to the Pacific Ocean, it is limited to regions with thorny shrubs and trees that offer nesting sites (Terres 1980).

Characteristic shrubs associated with habitat occupied by the cactus wren and within the coastal sage scrub community include California buckwheat (*Eriogonum*

fasiculatum), coastal sagebrush (*Artemisia californica*), several sages (*Salvia* spp.) and scattered shrubs approaching tree-size, such as laurel sumac (*Malosma laurina*), and lemonadeberry (*Rhus integrifolia*) (Garrett and Dunn 1981, Unitt 1984, Rea and Weaver 1990). Thickets of xeric vegetation may provide cover and thermal relief. The nest is also used as a roost site (Anderson and Anderson 1957).

Biology

Genetics: The variation in plumage patterns and characters are used to distinguish the subspecies of the cactus wren. Eight subspecies are recognized with the subspecies falling into roughly two groups the *affinis* group (peninsular forms) and *brunneicapillus* group (continental forms) (Proudfoot *et al.* 2000). The range of *C. b. cousei* is now geographically disjunct from interior desert populations as a result of urbanization of the corridor along the San Geronio Pass in Riverside County (Rea and Weaver 1990).

Diet and Foraging: The cactus wren forages on the ground and in low vegetation for insects and other small invertebrates, cactus fruits and other fruits, seeds and nectar (Bent 1968; Anderson and Anderson 1973). Fruits make up 15-20% of the annual diet, which is more than most North American wrens (Ehrlich, *et al.* 1988). Foraging behavior is often regulated by heat stress (Ricklefs and Hainsworth 1968), necessitating retreat from exposed sites into shade of shrubs and trees. The cactus wren generally forages on the ground, turning over fallen leaves and other debris in search of insects. It also searches bushes and probes tree bark housing insects. Foliage-gleaning may increase with insect abundance and habitat complexity (Proudfoot *et al.* 2000).

Daily Activity: The cactus wren exhibits year-long, diurnal activity. The species is not migratory (Zeiner, *et al.* 1990).

Reproduction: For the cactus wren, thickets of vegetation provide cover and shelter, and the nest, which is usually located in cactus, is used as a roost site as well as for breeding. The nest is usually built in cholla or other large, branching cactus, in yucca, or in a stiff-twigged, thorny shrub or small tree. The nest is an intricate, woven cylinder, usually placed horizontally 1.2 to 1.5 meters (4-5 feet) above the ground (Anderson and Anderson 1957). The large, globular chamber of the nest is about 18 centimeters in diameter with a tunnel-shaped passageway about 9 centimeters in diameter with as much as 30 centimeters between the back wall of the nest chamber and the entrance opening. The mouth of the entrance is usually about 7 centimeters above the base of the chamber. Because the passageway is too small to admit a flying bird, a doorstep or perch is required near the entranceway (Proudfoot *et al.* 2000). It breeds from March into June. The clutch size is 4-5 eggs, with a range of 3-7 eggs (Harrison 1978). Two broods per season is common. Incubation is 15-18 days, by the female only (Anderson and Anderson 1960). The altricial nestlings fledge at 17-23 days, with an average of 21

(Hensley 1959, Anderson and Anderson 1960). The young may return to roost in the nest after fledging. The young become independent at about 1 month after leaving the nest and sometimes the young help feed the young of later broods (Harrison 1978).

Survival: Anderson and Anderson (1973) report an overall adult survival rate of 50.6% during a six year study. One banded adult was retrapped when it was 4 years old (Terres 1980).

Dispersal: The species is generally considered to have low dispersal capabilities but there is little information available (Ogden Environmental and Energy Services 1993). In Arizona, of 55 nestlings banded, 41 dispersed from the natal site by 45 days postfledging. Males remain near the natal site, usually dispersing only as far as parental territorial behavior dictated (Proudfoot *et al.* 2000).

Socio-Spatial Behavior: The home range may be the same as the territory (Anderson and Anderson 1963). The average territory was 1.9 hectares (4.8 acres), varying from 1.2-2.8 hectares (2.9-6.9 acres), in Arizona (Anderson and Anderson 1973). The cactus wren may maintain its territory year-round (Anderson and Anderson 1963).

Community Relationships: Domestic cats, roadrunners, snakes, and loggerhead shrikes prey on adults and nestlings (Anderson and Anderson 1973). Austin, *et al.* (1972) observed nestling predation by gopher snakes and whipsnakes. Frequent interactions with curve-billed thrashers have been reported by Anderson and Anderson (1963), including destruction of cactus wren roosting nests by thrashers.

Known Distribution

The cactus wren is a resident species from southern California south to southern Baja California, southern Nevada, southwestern Utah, western and south central Arizona, southern New Mexico, and central Texas south to Mexico (Terres 1980).

Zeiner, *et al.* (1990) summarize the distribution, abundance, and seasonality of the cactus wren in California as follows. It is a locally common resident in the Mojave and Colorado deserts, north from the Mexican boundary to Inyo and Kern counties. The coastal race is found in arid parts of westward-draining slopes from San Diego County northwest to Ventura County. Historically, cactus wrens within coastal areas were found on the coastal slopes and lowlands of southern California in arid and semiarid regions with abundant cacti (Grinnell 1898, Grinnell and Miller 1944 Unitt 1984). As early as 1944, authorities noted that loss of habitat had greatly reduced the historic range of this species (Grinnell and Miller 1944).

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum and available literature. There are five occurrences of cactus wren in the Wash Planning Area; all occurrences are found in the San Bernardino County Natural History Museum database. All of the occurrences occur within either intermediate or mature Riversidean alluvial fan sage scrub in the eastern half of the Wash Planning Area.

Special Biological Considerations

The cactus wren is highly associated with cactus thickets in coastal sage scrub and is reliant on cacti for nesting, breeding and foraging. This species has an affinity for cholla cactus as a nesting and roosting site and this plant species is very important to its survival. (Bailey 1922; Grinnell and Miller 1944; Bent 1968; Anderson and Anderson 1973; Root 1988). The recommendations for protecting the cactus wren includes protection and maintenance of large blocks of coastal sage scrub through fire suppression (Rea and Weaver 1990).

Cactus wrens build four to six nests within their territories and thus enumerating the number of nests within an area is not a representative method for sampling population size (Anderson and Anderson 1973).

The cactus wren has been documented to have significant differences in clutch size, breeding success, and the timing of clutch initiation between years (Marr and Ratt 1983). These differences appear to be related to the annual differences in the abundance and emergence of the major food of the nestlings. Long-term temperature patterns may provide a predictor for high prey populations (Marr and Raitt 1983).

A flowchart was developed for the habitat suitability model for the cactus wren (Short 1985). Suitable habitat is evaluated as including: arid savanna, open thorn forest, or semi-desert cactus and deciduous tree cover types in southwestern United States; a block of appropriate habitat that is at least 0.4 hectare in area; habitat structure that provides potential nest sites 0.9 to 4.3 meters above ground; types of vegetation that vary in utility as nest sites for the cactus wren; and habitat should include a density of mid-story vegetation that may modify the utility of habitats for cactus wrens.

Threats to Species

Continued threats to the cactus wren include habitat loss and fragmentation from urbanization and agricultural development. Domestic cats, roadrunners, snakes, and loggerhead shrikes prey on adults and nestlings (Anderson and Anderson 1973). Cactus wrens that are confined to isolated patches of habitat in urbanizing areas are

subject to increased levels of predation pressures as larger predators are replaced by greater population levels of smaller predators and domestic animals. This species is especially vulnerable to stochastic events, especially wildland fires. Because of its narrow habitat requirements, sedentary behavior, and low dispersal characteristics, cactus wrens are subject to loss by fires and, if they disperse, may not find suitable habitat to survive. Intense fires may actually kill cactus plants and eliminate habitat for the cactus wren. As a result of competition from invasive plant competition, grazing, weather patterns and other natural and human-influenced disturbances, the reestablishment of cactus patches essential to this species may take many years. An increasing pattern of habitat fragmentation and isolated populations also diminishes the dispersal ability and inter-population connections of the cactus wren and reduces the overall genetic viability of the species (Ogden Environmental and Energy Services 1993).

3.2.8 Loggerhead Shrike

SPECIES DESCRIPTION

Status

The loggerhead shrike is a California Special Concern species and is listed federally by the U.S. Fish and Wildlife Service as a Bird of Conservation Concern. This species is not state or federally-listed as threatened or endangered.

Habitat and Habitat Associations

The loggerhead shrike is known to forage over open ground within areas of short vegetation, pastures with fence rows, old orchards, mowed roadsides, cemeteries, golf courses, riparian areas, open woodland, agricultural fields, desert washes, desert scrub, grassland, broken chaparral and beach with scattered shrubs (Unitt 1984; Yosef 1996). Individuals like to perch on posts, utility lines and often use the edges of denser habitats (Zeiner, *et al.* 1990). In some parts of its range, pasture lands have been shown to be a major habitat type for this species, especially during the winter season (Yosef 1996) and breeding pairs appear to settle near isolated trees or large shrubs (Yosef 1994). The highest density occurs in open-canopied valley foothill hardwood, valley foothill hardwood-conifer, valley foothill riparian, pinyon-juniper, juniper, desert riparian, and Joshua tree habitats and it occurs only rarely in heavily urbanized areas, but may be found in open cropland (Zeiner *et al.* 1990). In many regions, indices of the loggerhead shrike abundance correlate with the percentage of pastureland available (Gawlik and Bildstein 1993). In the Mojave Desert, the loggerhead shrike was observed more often in urban settings than other predatory bird species occurring there (Knight *et al.* 1999). In the mid-west the habitat use of the shrike is defined as savannah habitat at the

landscape scale but at the fine-scale, sites used by shrikes were characterized by tall, sparse, structurally heterogeneous herbaceous vegetation with high standing dead plant cover and low litter cover (Michaels and Cully 1998). The tree and shrub density did not differ between sites used and not used by shrikes (Michaels and Cully 1998).

Biology

Genetics: The endangered San Clemente shrike, *L. L. mearnsi*, shows only 60% of the genetic variation of the mainland shrike population even though individuals of the subspecies *gambeli* visit the island annually; it has been concluded that the island population has evolved sufficient genetic independence to justify ongoing conservation efforts (Mundy *et al.* 1997a). Strong structure is apparent in the cytochrome b mtDNA (mitochondrial DNA) sequence variations defining four haplotypes corresponding to the four subspecies (Mundy *et al.* 1997b). The similarity in haplotypes between populations from Saskatchewan and southern California suggests a post-glacial northern range expansion of the species (Mundy *et al.* 1997b).

Diet and Foraging: The loggerhead shrike foraging habitat includes open landscapes characterized by well-spaced, often spiny, shrubs and low trees, usually interspersed with short grasses, forbs, and bare ground, scrub lands, steppes, deserts, savannas, prairies, agricultural lands and some suburban areas (Yosef 1996). For foraging habitat, they appear to favor areas with fence lines and utility lines and poles for perching (Yosef 1996). In suboptimal foraging habitat areas, where grass is tall and dense, their foraging success is not affected, however their foraging methods are altered and include more hovering, more flights, and frequent changes in perches, generally more energetically expensive behaviors and thus larger prey items are taken (Yosef and Grubb 1993).

Individuals of the loggerhead shrike perch to search for prey which include large insects, small mammals, amphibians, reptiles, fish and invertebrates and they use impaling as a means of handling prey (Zeiner *et al.* 1990). Shrikes primarily subsist on large ground-dwelling insects and do not seem to require water (Miller and Stebbins 1964). Shrikes have been shown to be able to consume toxic insects by impaling and allowing them to “age” which apparently rids the then dead prey of the toxic chemical (Yosef and Whitman 1992).

Nestling shrikes have been successfully reared in captivity to create a model of feeding for the endangered San Clemente shrike. The artificially incubated, hatched and reared chicks were most successfully fed a varied diet consisting of mouse pups, egg and insects, which more closely resembles a diet provided in the wild than other artificial diets (Kuehler *et al.* 1993).

Daily Activity: The loggerhead shrike is a yearlong, diurnally active species (Zeiner *et al.* 1990). It spends approximately 80% of its day perched but will spend more time in flight in suboptimal foraging habitat areas (Yosef and Grubb 1993).

Reproduction: In an Idaho sagebrush (*Artemisia tridentata*) rangeland community, most loggerhead shrike nests (65%) were constructed in sage brush although bitterbrush and greasewood were also used frequently (Woods and Cade 1996). Nesting occurs in branches up to 4.5 meters above the ground frequently in a shrub with thorns or with tangled branching habits (Zeiner *et al.* 1990; Yosef 1996). Nests or nest materials are often reused in subsequent years (Yosef 1996). Height of nest shrubs average 162 centimeters and the mean height of nests was 79 centimeters although success of the nesting attempt did not appear to be related to the location of the nest but was more related to stochastic events such as predation and weather (Woods and Cade 1996). Porter *et al.* (1975) obtained reproduction data for a pair in shortgrass prairie habitat of Colorado. Mean nesting height was 2.03 meters, mean clutch size was 6.4 eggs, no double broods were observed although re-nesting was common if the nest failed (Porter *et al.* 1975). Throughout its range, the mean clutch size for the species is 5.4 eggs per nest with a range of 1 to 9 eggs (Yosef 1996). Loggerhead shrikes exhibit a latitudinal and longitudinal cline in clutch size with larger clutches at higher latitudes and farther west (Yosef 1996).

The parent shrikes may induce the young to fledge from the nest earlier than normal in order to avoid predation (Woods 1993). This may be due to the high predation rate on loggerhead shrike nestlings.

Loggerhead shrikes have been successfully hand-reared in captivity, will breed in captivity and can then be successfully released into the wild (Cade 1992).

Survival: The average nesting success of the loggerhead shrike, measured as the % of nests in which at least one young fledges, is 56% (Yosef 1996). The large clutch size and relatively high rate of hatching success, potentially enables the loggerhead shrike to produce large numbers of offspring, although many young are lost through brood reduction and predation (Yosef 1996). Predation has been calculated to account for 52% of all nest failures and adverse weather accounts for 33% (Porter *et al.* 1975).

Dispersal: Juvenile dispersal of the loggerhead shrike has been measured at around 12 to 14.7 kilometers from the natal site with adults dispersing a mean distance of 2.7 kilometers (Yosef 1996; Collister and De Smet 1997). Movement patterns of the shrike indicate that they disperse preferentially along connecting corridors of vegetation rather than between equally sized isolated patches of habitat (Haas 1995).

Socio-Spatial Behavior: In those geographic locations where the species is a year-round resident, the loggerhead shrike usually lives in pairs on permanent territories (Yosef 1996). For populations that are migratory a territory is defended through the non-breeding season and some pairs spend the entire year in a single territory (Miller 1931; Smith 1973). Outside the breeding season, the mates may defend neighboring territories, which are coalesced at the beginning of the nesting season (Yosef 1996). Miller and Stebbins (1964) observed large territories of 12-16 hectares while Yosef (1996) sites a mean territory size of 8.5 hectares. Territories in California are jointly defended by pairs during the breeding season, but during the fall these pairs disband and defend separate, although often adjacent, winter territories (Yosef 1996).

Community Relationships: Loggerhead shrikes interact with many other avian species with which they share habitat; shrikes often dominate these interactions (Yosef 1996). Shrikes may also compete with the fire ant for food sources which may cause declines in the population of shrikes (Grisham 1994).

Known Distribution

Throughout most of the southern portion of its range, the loggerhead shrike is a resident except as described by Terres (1980;Yosef 1996). The northern populations are migratory (Yosef 1996). The species nests from southern Canada through the Great Basin and California, to Baja California, Mexico and the Gulf coast (Terres 1980). Specifically, in western North America, the species breeds from southeastern Alberta, western Montana, northwest Wyoming, southern Idaho, south-central Washington, eastern Oregon, and California south to southern Baja California. In Central North America, it breeds from southern Saskatchewan and southwest Manitoba, North Dakota, and portions of southern Minnesota, eastern Iowa, northwest and southeast Missouri and northern Arkansas, south through Louisiana, Texas, New Mexico, and Arizona and through Mexico to north Sinaloa and Oaxaca. In eastern North America, it breeds in southern Wisconsin, and from southeast Illinois and southwest Ohio south to the Gulf Coast and from eastern West Virginia and all but the eastern portions of both Virginia and North Carolina south to the Gulf Coast and all but the extreme southern part of Florida (Yosef 1996).

Wintering grounds are found in the southern portion of the breeding range and further south into Mexico (Terres 1980). The northern populations are migratory and most winter from northern California, northern Nevada, northern Utah, central Colorado, southern and eastern Kansas, western Missouri, northern Kentucky, and northern Virginia south through the southern United States and in Mexico south throughout the breeding range (Yosef 1996).

It is difficult to document and compare historic and current distributions of the loggerhead shrike because the number of observers and level of survey effort has increased dramatically. Clearing of virgin forests and replacement by open farmlands may have allowed the species to become widely distributed before the beginning of the twentieth century. Many of the habitats in which this species breeds are seral stages of vegetation, although deserts, shrub steppes, and southern savannas may represent the historic Core Areas of its distribution (Yosef 1996).

In California, the species is found throughout the foothills and lowlands of California as a resident (Zeiner *et al.* 1990). Winter migrants are found coastally, north of Mendocino county (Zeiner *et al.* 1990). The loggerhead shrike seems to have always been most abundant in the southern and western portions of its range (Cade and Woods 1997).

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum and available literature. There are six occurrences of loggerhead shrike; all occurrences are found in the San Bernardino County Natural History Museum database. All of the occurrences occur within either intermediate or mature Riversidean alluvial fan sage scrub with the exception of one location within disturbed habitat. The occurrences are distributed throughout the Wash Planning Area.

Special Biological Considerations

Smyth and Coulombe (1971) report that the loggerhead shrike does not drink water up to ambient air temperatures of 40°C. The loggerhead shrike has a basal metabolic rate that is lower than predicted for a passerine its size but is more similar to other raptor species (Cunningham 1979). Additionally, it has an extended thermoneutral zone from approximately 24 to over 36 degrees C (Weathers, *et al.* 1984). Cunningham (1979) concluded that as an animal that is a sit and wait predator, it benefits from having a reduced basal metabolic rate. The most metabolically expensive behavior of the bird is flight (Weathers, *et al.* 1984) which it uses rarely: it spends approximately 80% of its day perched but will spend more time in flight in suboptimal foraging habitat areas (Yosef and Grubb 1993). Management implications are that short grass areas are important to include for use by shrikes for hunting (Yosef and Grubb 1993).

In addition to using barbed wire for impaling food items, the loggerhead shrike has been observed using barbed wire to anchor and tear nest-lining materials (Burton 1999). Effects of protective fencing were found to result in higher abundance and species richness of birds, including the loggerhead shrike for which nesting was also found to be more frequent inside the fenced area. This increase in abundance may be related to an increase in abundance of seed and invertebrate food sources, and particularly for the shrike, an increase in reptile prey species (Brooks 1999).

Sites used by loggerhead shrikes did not differ with respect to military training disturbance, hay harvest, or the number of years since a site was last burned (Michaels and Cully 1998). Movement patterns of the shrike concluded that they disperse preferentially along connecting corridors of vegetation than between equally sized isolated patches of habitat (Haas 1995). Management for resident shrikes should include a patchwork of grassy habitats and sparsely vegetated bare areas at the scale of individual shrike territories (Gawlik and Bildstein 1993). Similarly, in southern Alberta, comparisons of occupied and unoccupied sites indicate that the availability of suitable habitat for breeding could be increased by management practices that increase the prevalence of grasslands (Prescott and Collister 1993; Telfer 1992). Woods and Cade (1996) found shrublands to be a very important habitat used by the shrike for nesting and that a contraction in both range and numbers of shrikes has occurred as the appropriate habitat has been lost.

Brooks and Temple (1990) used a model to demonstrate that a minimum of 5.5 fledglings per breeding pair per season was required for a stable population in Minnesota if the territory reoccupancy of 47% equals the adult survival and assuming that first year survival of 19% is correct. This appears to be an unrealistic model and the investigators found an actual rate of only 2.07 fledglings per breeding pair per season.

Since reproductive rates are potentially high in this species, it could expand its current numbers and range if the factors responsible for its recent decline can be identified and eliminated. In some studies, the reproductive success of loggerhead shrikes is relatively high and does not explain the recent decline in shrike populations (Gawlik and Bildstein 1990). Although much good habitat has been lost, enough remains unoccupied to accommodate a larger population (Yosef 1996). Management initiatives aimed at benefitting the loggerhead shrike should be directed toward increasing prevalence of medium and perhaps tall grass in favorable areas for populations. This can be achieved by controlling grazing and mowing of grasslands. It is also desirable to maintain brush along fence lines, scattered trees in pastures and fields and hedges as potential nest sites (Yosef 1996). The density of hunting perches affects the nutritional condition of shrikes (Yosef and Grubb 1992). Areas devoid of shrike territories have been noted to contain few fences and sparse trees, thus these areas may have been subthreshold economically because of insufficient foraging features for shrikes. The continuing declines of the loggerhead shrike may be at least partially due to withdrawal from parts of the species range in which man's activities have reduced the density of hunting perches (Yosef and Grubb 1992).

Key management priorities for the loggerhead shrike include: determine the migration routes, stopover and wintering areas and the susceptibility to human disturbance at

these locations; evaluate the dietary needs and how weather, season, land use, and biocides influence food availability; determine the mortality rates of fledged juveniles and adults throughout the annual cycle in different habitats; determine the degree of niche overlap between the loggerhead shrike and potential competitors to see whether shrike productivity is correlated with the presence or absence of these species (Yosef 1996).

Threats to Species

Despite its wide distribution, the loggerhead shrike is one of the few North American passerines whose populations have declined continent-wide in the recent decades (Yosef 1996). Terres (1980) cites that shrike are often killed by automobiles early in the morning. In one study, the pesticide DDE may have reduced an Illinois population through eggshell thinning (Anderson and Duzan 1978; Morrison 1979). Pesticide use (organochlorines, DDE, etc.) may have potentially reduced eggshell thickness and altered development (Yosef 1996). Displacement of habitat through urban development, the spraying of biocides, and competition with species that are more tolerant of human-induced changes may be resulting in population declines (Yosef 1996). The loggerhead shrike is thought to be generally tolerant to human harassment, although it will abandon nesting attempts if disturbed (Yosef 1996). A study of the effect of spraying the common fertilizer, sodium ammonium nitrate, on cattle pastures concluded that the foraging territories of shrikes increased on average to 138% of a control group and the survivorship of eggs, nestlings and fledglings as well as adults was reduced, and one territory was abandoned (Yosef and Deyrup 1998).

The loggerhead shrike was once widely distributed and common over most of North America, occupying an exclusive breeding range with no other shrikes (Cade and Woods 1997). Although it occurs in a wide variety of plant associations, this shrike is generally found in landscapes characterized by widely spaced shrubs and low trees interspersed with short grasses, forbs, and bare ground, habitat conditions which are currently being developed (Cade and Woods 1997). Recently, Christmas bird count data and Breeding Bird Survey data have revealed an overall downward trend across the continent that appears to be related to alterations in habitat structure and loss of habitat as well as the loss of pasture lands and increase in intensive row-crop agriculture (Cade and Woods 1997; Prescott and Collister 1993; Telfer 1992; Gawlik and Bildstein 1993; Smith and Kruse 1992). Most populations along the coastal plains of southern California have been displaced by urban development, although the subspecies occupying the region (*L. l. gambeli*) is not yet in danger of extirpation (Morrison 1981).

The loggerhead shrike may suffer population declines due to the presence of the fire ant. Studies have looked at changes in the shrike's winter habitat and found that, in

addition to changes in land use, the shrike's decline in particular counties in the southern U.S. are directly correlated with an increase in fire ants in the area. The shrike and fire ant are direct competitors for food sources. Both feed on invertebrates, reptiles, and small mammals. It is possible that fire ant control could benefit the loggerhead shrike in those areas where they co-occur (Grisham 1994). Currently there is a known location of the fire ant in Western Riverside County; however, the distribution of the fire ant within San Bernardino County and the Wash Planning Area is unknown.

The San Clemente loggerhead shrike appears to be threatened by the introduction of exotic species which have altered the ecosystem of San Clemente Island including loss of suitable habitat and increased predation of nests and adults (Scott and Morrison 1990).

3.2.9 Southern California Rufous-crowned Sparrow

SPECIES DESCRIPTION

Status

The southern California rufous-crowned sparrow is a California Special Concern species. This species is not state or federally-listed as threatened or endangered.

Habitat and Habitat Associations

Southern California rufous-crowned sparrows are found on moderate to steep, dry, grass-covered hillsides, coastal sage scrub, and chaparral and often occur near the edges of the denser scrub and chaparral associations. Preference is shown for tracts of California sagebrush (*Artemisia californica*) (Collins 1999). It also occurs in grass that grows as a successional stage following brush fires and sparse chaparral recovering from a burn as well as the edges of tall chaparral (Unitt 1984, Collins 1999) and may be found in open shrubland in valley foothill hardwood-conifer savannah and open chaparral (Verner and Boss 1980). Optimal habitat consists of sparse, low brush or grass and hilly slopes preferably interspersed with boulders and outcrops (Willet, 1912, 1933; Grinnell 1915, 1926, Grinnell and Miller 1944; Bent 1968; Pulliam and Mills 1977; Phillips, *et al.* 1983; Unitt 1984; Ehrlich, *et al.* 1988; Root 1988). The species may occur on steep grassy slopes without shrubs if rock outcrops are present (Zeiner *et al.* 1990). Some observers have noted a preference for south-facing or west-facing slopes and an affinity for California sagebrush over other vegetative types (Barlow 1902, Grinnell 1915, Grinnell and Miller 1944, Bent 1968; Root 1988). It is uncommon on the lower slopes of the western Sierra Nevada, and on Santa Cruz Island (Grinnell and Miller 1944). It is most numerous in the western portion of its range in California (Zeiner *et al.* 1990). It is generally absent from dense, unbroken stands of coastal sage scrub and

chaparral. The elevation range in California has been recorded as 60 to 1,400 meters (Collins 1999).

The physical and vegetative characteristics of the Southern California rufous-crowned sparrow have been quantified by Collins (1999). The results have confirmed that this species prefers moderate west, south-, and east-facing slopes vegetated with low, fairly open cover of shrubs and grass. Most of the inhabited sites (89%) were on slopes of 15 to 60 degrees. Almost half of the sites were on moderate slopes (30 to 45 degrees). Rock outcrops were present at 61% of the occupied sites. Shrub and grass were the dominant cover types with shrubs averaging 50% cover and grass averaging 29%. Shrub height was generally low in this study, averaging 0.83 meter. The dominant overstory shrubs associated with the Habitats used by this species include California sagebrush, purple sage (*Salvia leucophylla*), black sage (*S. mellifera*), California encelia (*Encelia californica*), coyote brush (*Baccharis pilularis*), mock heather (*Ericameria ericoides*), deer weed (*Lotus scoparius*), giant rye (*Leymus condensatus*), and buckwheat (*Eriogonum* sp.).

Biology

Genetics: The rufous-crowned sparrow has been placed in several different genera depending on the past nomenclatural and taxonomic conventions and decisions: *e.g.*, *Ammodramus*, *Zonotrichia*, *Peucaea*, and *Aimophila*, where it is currently assigned (Collins 1999). The Southern California rufous-crowned sparrow is similar to the nominate *ruficeps*, except the wing and tail are longer, the bill is smaller, the underparts brown with a grayish wash, and the upper parts are rufous-brown with a grayish-buff streaking (Collins 1999).

Diet and Foraging: The rufous-crowned sparrow forages on the ground in herbage and in litter beneath shrubs, gleaning from ground and foliage; the species also gleans foliage of live oak, foraging predominantly on insects during the breeding season and including seeds, grasses, and forb shoots at other times of the year (Verner and Boss 1980; Bent 1968). Generally the diet is poorly known - it appears to vary with season, locality, and availability. It may eat more insects during the spring and summer and more seeds during the winter (Collins 1999).

Daily Activity: All Southern California rufous-crowned sparrow activities are focused on and around the ground, usually in the area of dense vegetative cover (Grinnell and Miller 1944; Bent 1968, Pulliam and Mills 1977; Root 1988). The species exhibits year-long, diurnal activity (Zeiner, *et al.* 1990). Males typically sing at all hours of the day with a peak activity in the early morning and late afternoon (Collins 1999).

Reproduction: The rufous-crowned sparrow breeds and feeds on steep, dry, herbage-covered hillsides with scattered shrubs and rock outcrops. Southern California rufous-crowned sparrows are relatively secretive, seeking cover in shrubs, rocks, grass and forb patches, concealing their nest on the ground at the base of a grass tussock or shrub or about 1 to 3 feet above the ground (Terres 1980; Verner and Boss 1980). The nest is cuplike and made of twigs, bark strips, grasses, and is lined with hair of deer, horses, and grasses (Terres 1980).

The rufous-crowned sparrow breeds from mid-March to mid-June with a peak in May. The egg dates for California are 11 March to 15 June with most occurring in April and May. The species is known to be monogamous however breeding territories may occur in groups (Pemberton 1910). The pairs are maintained throughout the year (Collins 1999). The clutch size is 2-5 eggs, and is usually 3 or 4 eggs. Incubation is by the female only, but the altricial young are tended by both parents (Harrison 1978). Incubation lasts about 11 to 13 days (Collins 1999). The nesting period is estimated to last 8 to 9 days and at nest departure, the young are not completely feathered and their wings and tails are only partially grown. Fledglings are incapable of flight upon nest departure and are usually found either moving through low vegetation or hopping or running on the ground under protective cover of the vegetation (Collins 1999). Seasonal fecundity estimates for a population in southern California were 3.98 and 4.86 young/pair/season in 1996 and 1997 respectively (Collins 1999).

Survival: The oldest individual of the rufous-crowned sparrow that has been reported is 3 years, 2 months (Klimkiewicz and Fitcher 1987).

Dispersal: The Southern California rufous-crowned sparrow is not migratory. There may be some movement up slope during the postbreeding period to 1220 meters (4000 feet) in the western Sierra Nevada (Gaines 1977). It generally remains on or near the preferred breeding Habitat throughout the fall and winter. In the San Gabriel Mountains, individuals or pairs were observed during the fall in or near most of the territories used for breeding during the breeding season that had just finished. There may be limited postbreeding wandering of the young and adults into nearby Habitats that are not used for breeding (Collins 1999).

Socio-Spatial Behavior: Home range of the Southern California rufous-crowned sparrow, estimated from nesting density, was about 1.5 hectares (3.7 acres) in southern California chaparral (Cody 1974). In Arizona oak woodland, Balda (1969, 1970) reported six pairs and 11 pairs per 40 hectares (100 acres). In southern California coastal sage scrub, the territory size averages 2.0 acres with a range from 1.2 to 3.2 acres (Bent 1968). The species is not gregarious and is generally found in groups composed of no greater than five or six (Bent 1968) and apparently exist in scattered metapopulations across patchy landscapes. In southern California, territorial males are

closely spaced in coastal sage scrub and more widely spaced in regrown (3-5 years post-fire) hard chaparral (Collins 1999).

Community Relationships: Eggs and nestlings of the Southern California rufous-crowned sparrow are preyed upon by snakes and small mammals (Bent 1968). Friedmann (1971) reported the first record of cowbird parasitism in this species. The Southern California rufous-crowned sparrow may occur in family groups postbreeding (Ehrlich, *et al.* 1988).

Known Distribution

The rufous-crowned sparrow, including all subspecies, is largely a resident species and occurs in central California, north-central Arizona, southwestern New Mexico, southeastern Colorado, northwestern and central Oklahoma, south discontinuously to southern Baja California and Mexico. The species occurs throughout much of the southwestern United States and Mexico but the range is often discontinuous with numerous small, isolated populations (Collins 1999). East of the Rocky Mountains, it winters from central and southern Oklahoma to northern Texas and south into Mexico (Terres 1980).

The current range and distribution of the Southern California rufous-crowned sparrow subspecies is extremely restricted to a narrow belt of semiarid coastal sage scrub and sparse chaparral from Santa Barbara south to the northwestern corner of Baja California. (Todd, 1922, Grinnell, 1926, Grinnell and Miller 1944, Bent 1968, Zeiner, *et al.* 1990; Unitt 1984). It is generally resident throughout its range, and no true migratory movements have been recorded. Limited movements to lower elevations have been reported during especially severe winters (Collins 1999).

The conversion of large areas of coastal sage scrub for urban and agricultural developments have made this species more locally restricted in various southern California counties (Los Angeles, Orange, Riverside, San Diego, and San Bernardino counties) (Collins 1999).

Status in Wash Planning Area

Data reviewed includes the CNDDB, the USFWS database, the San Bernardino County Natural History Museum and available literature. There are nine occurrences of rufous-crowned sparrow in the Wash Planning Area; all occurrences are found in the San Bernardino County Natural History Museum database. Most of the occurrences occur within Riversidean alluvial fan sage scrub (three in pioneer, three in intermediate, one in intermediate/mature, and one in mature) with two of the species occurrences in areas

mapped as developed and disturbed. The rufous-crowned sparrow occurrences are distributed throughout the Wash Planning Area.

Special Biological Considerations

Although knowledge of Southern California rufous-crowned sparrows in the Wash Planning Area appears to be limited, the species appears to have relatively broad habitat preferences and a scattered distribution (Garrett and Dunn 1981).

Given the available information, this species apparently would benefit from steep slope preservation and maintenance of open edge conditions of coastal sage scrub that perpetuate herbaceous (grass and forb) elements. The limited use of prescribed fires may provide the disturbance that enhances foraging areas for this species.

Bolger *et al.* (1997) studied the 20-most common bird species within a 260 km² area of coastal San Diego County in relation to edge/fragmentation sensitivity. Southern California rufous-crowned sparrow was found to be one of four species whose abundance is most reduced by presence of edges/fragmentation.

Declines in the populations in southern California suggest that the larger, interconnected blocks of open scrub Habitat suitable for maintaining this species over the long term should be preserved, particularly in areas under intense pressure from urban and irrigated or mechanized agricultural developments. Because this species can apparently tolerate a moderate amount of disturbance from prescribed burning and moderate levels of grazing, and because it usually inhabits moderate to steep slopes, management of its populations elsewhere in California and in other southwestern states may not be needed (Collins 1999). Moderate grazing and trampling by cattle on canyon slopes may benefit the rufous-crowned sparrow by opening up dense shrub Habitats (Collins 1999).

Threats to Species

The loss of coastal sage scrub for agriculture and urban development has reduced the available Habitat for the Southern California rufous-crowned sparrow (Bent 1968; Unitt 1984). Other stressors include a range of avian, mammalian and reptilian predators, both native and domestic, that find the ground-nesting habit of this bird as an easy target (Bent 1968). Long term fire suppression since the turn of the twentieth century may also have contributed to the reduction in numbers in California by allowing the chaparral and coastal sage scrub Habitats to grow into dense, decadent stands. It is only a rare host to brood parasitism of the brown-headed cowbird (*Molothus ater*) (Friedmann 1971, Collins 1999).

3.2.10 Los Angeles Pocket Mouse

SPECIES DESCRIPTION

Status

The Los Angeles pocket mouse is a California Special Concern Species. It is not state- or federally-listed as threatened or endangered.

Habitat and Habitat Associations

Because the habitat associations in the project area are similar to the San Bernardino kangaroo rat in the Wash Planning Area as discussed earlier, the following habitat description draws heavily upon the San Bernardino kangaroo rat habitat assessment conducted by URS (2003a) for the San Bernardino Valley Water Conservation District.

Habitat of the Los Angeles pocket mouse has never been specifically defined, although Grinnell (1933) indicated that the subspecies “inhabits open ground of fine sandy composition” (cited in Brylski *et al.* 1993). This observation is supported by others who also state that the Los Angeles pocket mouse prefers fine, sandy soils and may utilize these soil types for burrowing (*e.g.*, Jameson and Peters 1988). This subspecies may be restricted to lower elevation grassland and coastal sage scrub (Patten *et al.* 1992).

Within the Wash Planning Area, the Los Angeles pocket mouse is most likely to occupy the Riversidean alluvial fan sage scrub vegetation within the Santa Ana River floodplain, which is comprised of three primary seral stages of alluvial fan sage scrub: pioneer, intermediate, and mature phases.

Pioneer phase alluvial fan sage scrub is the initial colonizing stage where recent scouring and flood events have occurred. This phase is characterized by very sparse distributions of subshrubs dominated by bristly goldenaster (*Heterotheca sessiflora* spp. *echiodes*) and Scalebroom (*Lepidospartum squamatum*). Because of typically recent scouring and flooding and the lack of a well-developed vegetation community, pioneer phase alluvial fan sage scrub is less suitable for the Los Angeles pocket mouse compared to intermediate alluvial fan sage scrub. However, these areas probably are integral to the overall habitat system and life history of the Los Angeles pocket mouse with regard to temporary use and dispersal and potential succession to more suitable habitat over time.

Intermediate phase alluvial fan sage scrub is comprised mainly of subshrubs such as California buckwheat (*Eriogonum fasciculatum*), brittlebush (*Encelia farinosa*), yerba santa (*Eriodictyon tricholax*), Spanish bayonet (*Yucca whipplei*), deerweed (*Lotus*

scoparius), valley cholla (*Opuntia parryi*), and coastal prickly-pear (*Opuntia littoralis*). Intermediate phase alluvial fan sage scrub typically occurs on terraces above scoured channels. Intermediate phase alluvial fan sage scrub is considered the highest quality habitat for the Los Angeles pocket mouse because this phase retains open, sandy areas favored by the species. Intermediate phase alluvial fan sage scrub is expected to support the highest densities of the Los Angeles pocket mouse of the three primary seral stages.

Mature phase alluvial fan sage scrub typically occurs on higher terraces away from the active flood channel that have not been subjected to flooding and scouring for many years. The mature phase is characterized by large woody species such as California juniper (*Juniper californicus*), Spanish bayonet, chamise (*Adenostoma fasciculatum*), holly-leaved cherry (*Prunus ilicifolia*), sugarbush (*Rhus ovata*), redberry (*Rhamnus crocea*), hoaryleaf ceanothus (*Ceanothus crassifolius*). Because the mature phase is characterized by relatively dense vegetation with few sandy openings, it is considered less suitable for the Los Angeles pocket mouse than the intermediate phase. However, as with the pioneer stage, the Los Angeles pocket mouse may occasionally use mature alluvial fan sage scrub (as well as adjacent upland areas) and it may serve an important refugia function during large flood events.

While other vegetation associations may support the Los Angeles pocket mouse, like other heteromyid species, it probably prefers sparsely vegetated habitats. For another subspecies, the Pacific pocket mouse (*P. l. pacificus*), evidence indicates that mice avoid dense grass cover because of difficulty locomoting and finding seeds (M. Pavelka 1998-99; cited in Spencer and Schaefer 2000). Therefore, as with the San Bernardino kangaroo rat, areas in the Wash Planning Area mapped as chaparral, chaparral/non-native grassland and non-native grassland were not included as habitat.

Soil characteristics also must be appropriate for a site to support the Los Angeles pocket mouse. The large majority of the Wash Planning Area Plan Area supports Soboba stony loam sand, with a substantial component of psamments (sands) and fluvents within the historic active channels (NRCS Soil Survey Geographic [SSURGO] Database 2004). There are smaller pockets of Cieneba sandy loam and Hanford coarse sandy loam. All these soils probably are suitable for the Los Angeles pocket mouse either as burrowing habitats, where the soils have been consolidated by vegetation as occurs in intermediate alluvial fan sage scrub, or as foraging habitat in pioneer alluvial fan sage scrub.

Biology

Very little biological information is available specifically for the Los Angeles pocket mouse (*P. l. brevinasus*). Therefore, the common name used in this section where appropriate, is the little pocket mouse, which refers to the full species *P. longimembris*.

Genetics: The Los Angeles pocket mouse (*P. l. brevinasus*) is one of 16 subspecies of the little pocket mouse (Williams *et al.* 1993). The diploid number of chromosomes for the little pocket mouse is 56. There are no published data at this time of the genetic structure and diversity of the little pocket mouse. Genetic studies of different subspecies and populations of the little pocket mouse utilizing mitochondrial DNA (mtDNA) and nuclear microsatellites techniques currently are being conducted by Dr. James Patton of UC Berkeley. While the focus of the Patton study is on the recovery of the endangered Pacific pocket mouse (*P. l. pacificus*), the results of this study should be very relevant and important to this HCP.

Diet and Foraging: Like other heteromyids (pocket mice, kangaroo rats, and kangaroo mice), little pocket mice primarily are granivores (seed eaters). However, the little pocket mouse may specialize more on grass seeds than do other pocket mice and kangaroo rat species. For example, Meserve (1976) offered a variety of seeds to Pacific pocket mice (*P. l. pacificus*) “cafeteria” style and found that they strongly selected the seeds of ripgut grass (*Bromus [rigidus] diandrus*), foxtail chess (*Bromus madritensis* ssp. *rubens*), and purple needlegrass (*Nassella [Stipa] pulchra*). Forbs and perennial seeds selected (at least 26-50% consumed) included cudweed aster (*Lessingia [Corethrogyne] filaginifolia*), cotton-batting plant (*Gnaphalium [chilense] stramineum*), and rosin-weed (*Osmadenia [Calycadenia] tenella*). Whether the Los Angeles pocket mouse selects seeds of these species similar to the Pacific pocket mouse is unknown. All these plant species, except perhaps rosin-weed, are common in the range of the Los Angeles pocket mouse.

Beyond specialization on seeds, little is known of the foraging behavior of the Los Angeles pocket mouse. However, Reichman and Price (1993) provide a comprehensive treatment of heteromyid foraging that probably is generalizable to the Los Angeles pocket mouse. Pocket mice possess external, fur-lined cheek pouches that promote collecting and caching of seeds either in scatter- or larderhoards, but it is not known which pattern the Los Angeles pocket mouse exhibits in the wild. However, laboratory tests by Lawhon and Hafner (1981; cited by Price and Jenkins 1986) found that little pocket mice cached seeds in larderhoards more often than two kangaroo rat species (*Dipodomys merriami* and *D. panamintinus*). Price and Jenkins (1986) suggest that larderhoarding by little pocket mice may be related to their dormancy (torpor) in the winter.

Pocket mice (*Chaetodipus*, *Perognathus*) tend to forage under shrub and tree canopies, or around rock crevices, in contrast to kangaroo rats (*Dipodomys* ssp.) and kangaroo

mice (*Microdipodops* spp.) which tend to forage in more open areas (Reichman and Price 1993). Brown and Lieberman (1973) observed the little pocket mouse foraging around clumps of vegetation. Kenagy (1973) also observed that little pocket mice rarely occurred in the open and spent most of their time in or near bushes. The reliable occurrence of different species in different microhabitats is well documented, but reasons for these microhabitat preferences are not well understood (Reichman and Price 1993). Factors such as inter-specific competition, foraging economics, and predation risk probably are important factors in microhabitat selection, but the mechanisms and functions of such selection are not known.

Daily and Seasonal Activities: The daily activities of the Los Angeles pocket mouse have not been studied, but various studies of the little pocket mouse indicate that its daily activity patterns are similar to other heteromyid rodents (e.g., Kenagy 1973; O'Farrell 1974). Little pocket mice primarily are nocturnal, with an initial bout of surface activity within two to four hours after sunset and then declining activity throughout the night. In spring and summer, there may be a smaller bout of surface activity before sunrise (O'Farrell 1974).

Little pocket mice exhibit a distinct seasonal pattern in surface activity (Chew and Butterworth 1964; Kenagy 1973; O'Farrell 1974). During the colder months the little pocket mouse may enter into torpor and not engage in surface activity. For example, in a study of a rodent community in west-central Nevada, O'Farrell (1974) recorded little pocket mice on the surface beginning in April, with peak abundances in June and July. By August, surface activity was in decline and was almost absent in October. No surface activity was recorded from November to March. Likewise, Chew and Butterworth (1964) did not trap the otherwise common little pocket mouse during most of the fall and winter months in Joshua Tree in the Mojave Desert. Kenagy (1973) observed similar patterns in the Great Basin Desert, with peak surface activity occurring from May through August and little activity between October and March. Surprisingly, Kenagy recorded surface activity at surface temperatures as low as -10° Celsius. This pattern of seasonal activity is apparent with the Los Angeles pocket mouse in the Wash Planning Area. For example, a total of five individuals were trapped on two different grids at Lake Perris in June 1996, but no individuals were trapped on the same trap lines in October of the same year (Dudek & Associates, Inc. 1997). Kenagy (1973) observed that males emerged on the surface earlier than females after their dormant period.

Kenagy (1973) attributes the little pocket mouse's decrease in winter activity to an increase in the cost-benefit ratio of foraging. During the winter energy maintenance requirements increase while the availability of food decreases. At some point when surface conditions are very cold and food is scarce, the animal cannot meet its energy needs by foraging and thus must shut down surface activity to survive the winter.

During this period of dormancy, pocket mice survive on the food they have cached to their burrows.

Reproduction: As with other heteromyids, *P. longimembris* are not prolific breeders. In the laboratory Hayden *et al.* (1966) recorded typical gestation periods of 22-23 days. Females apparently are capable of breeding in their natal season and are reproductively active by as early as 41 days of age. In the wild, little pocket mice may produce one or two litters per year with typical litter sizes of 3-4 pups. Chew and Butterworth (1964) had few observations of reproduction in a population of the little pocket mouse in Joshua Tree, but reported pregnant females, males with testicular development, and very young animals in February through April. Kenagy (1973) found that males showed testicular enlargement within several weeks of emergence following the dormant period. Females showed evidence of vaginal activity (opening, swelling, and bleeding) shortly after emergence in the spring to September or October.

Survival: There are little data on survival in the wild in the little pocket mouse. It may live up to eight years in captivity (Edmonds 1972). In the wild, Chew and Butterworth (1964) recorded about 30% survival from one spring to the next in a population in Joshua Tree. They attributed this relatively high survival rate to the species' entering torpor during the cold months. Over three winters, Kenagy (1973) reported survival of 82%, 56%, and 36% from autumn to spring. In the year of highest survival, pocket mice were active all winter and the food supply was greater than the following two winters. In the following two winters, rainfall was below normal, presumably food supplies were scarce, and individuals entered dormancy. Kenagy's data indicate that dormancy is not a strategy to maximize survival, as Chew and Butterworth appear to suggest, but rather a strategy to minimize mortality. That is, when conditions support a low cost-benefit ratio of surface activity, survival is highest. When conditions are poor and the cost-benefit ratio of surface activity increases, dormancy provides the best opportunity to survive the winter.

Dispersal: A study of movement and dispersal by the Pacific pocket mouse on the Dana Point Headlands site in southern Orange County showed an average maximum distance moved of 19.7 meters, with a range of 4.0 to 87.0 meters (Spencer and Schaefer 2000). For adults the mean maximum distance was 26.4 meters and for young-of-the-year the mean distance was 18.9 meters. However, the Dana Point site is small and may limit the distance pocket mice may move compared to larger habitat areas. There are some data from MCB Camp Pendleton suggesting that juveniles may move up to several hundred meters between habitat patches in an "unconstrained system" (Spencer and Schaefer 2000). Trapping data from Chew and Butterworth at Joshua Tree indicate that the little pocket mouse shows high site fidelity from year to year. Of 19 individuals trapped in a second spring, 16 were trapped within two trap

stations (100 feet) of the previous year, and of these 16, eight were trapped one station away (50 feet) from the previous year.

Socio-Spatial Behavior: Heteromyids (pocket mice, kangaroo rats, and kangaroo mice) in general are asocial, solitary animals. Except during reproduction, they do not frequently engage in direct social encounters. Based on a trapping study in west-central Nevada, O'Farrell (1980) determined that little pocket mice home ranges overlapped during the peak breeding season of May through July, with a later peak in the second half of August. No overlap was observed when surface population numbers were low in April and September-November. In contrast to many other heteromyids, little pocket mice in this study showed more female-female range overlap than male-male overlap. O'Farrell (1980) characterizes the little pocket mouse as relatively more social than other heteromyids studied.

Crude estimates of home range size were made by Chew and Butterworth (1964) for the Joshua Tree population based on grid trapping data. They reported home range diameters of 38.7 meters to 85.4 meters, with an average of 64.3 meters. Circular home ranges based on these diameters would be 0.1 ha (0.25 acre) to 0.5 ha (1.2 acres), with an average of 0.3 ha (0.74 acre). In the Nevada desert, Maza *et al.* (1973) reported home ranges of females to be 0.5 ha (1.2 acres) to 3.1 ha (7.6 acres) and for males 0.3 ha (0.7 acre) to 1.9 ha (4.7 acres). Kenagy (1973) never trapped an individual little pocket mouse in more than one quadrat (each quadrat was 62.5 meters to the nearest quadrat) and he concluded that individuals moved much less than 50 meters during the night.

Population densities in the Chew and Butterworth (1964) study were 0.7 to 1.7 individuals/ha.

Community Relationships: The community ecology of heteromyid rodents, including kangaroo rats (*Dipodomys* spp.), pocket mice (*Perognathus* and *Chaetodipus* spp.) and kangaroo mice (*Microdipodops* spp.) is among the most studied aspect of this family's biology. Brown and Harney (1993) provide a comprehensive overview and attempted synthesis of this complex subject.

Arid grassland and desert environments support a surprising diversity of coexisting rodent granivores. The diversity and number of coexisting species varies depending on local conditions and the requirements of the constituent species. The Los Angeles pocket mouse in western Riverside County probably overlaps with at least four kangaroo rat species (*D. agilis*, *D. merriami*, *D. stephensi* and *D. simulans*), two other pocket mice (*Chaetodipus californicus* and *C. fallax*), and at least six native murids (*Peromyscus maniculatus*, *P. eremicus*, *P. californicus*, *Neotoma lepida*, *N. fuscipes*, and *Reithrodontomys megalotis*) that potentially compete for space and food resources.

Brown and Harney (1993) conclude that “the composition of these assemblages is not random. Instead it is determined by interactions of the species with the physical environment, with other kinds of organisms, and with other rodent species.” page 646. Generally, species that do coexist tend to occupy and exploit different microhabitats or niches or differ in their seasonality of resource exploitation.

Interspecific competition is an important component of the organization of heteromyid community structure. For example, competitive exclusion can result in nonrandom assemblages that partition the resources and habitats in the community. Other potential mechanisms of resource partitioning listed by Brown and Harney (1993) include habitat selection or restriction, independent adaptations, food partitioning and variable foraging efficiency, seed distribution, resource variability, predator-mediated coexistence, aggressive interference, and seasonality. It was noted above that little pocket mice tend to forage under and near shrubs and avoid open spaces that are more likely to be used by kangaroo rats (Brown and Lieberman 1973; Kenagy 1973).

Pocket mice and other heteromyid rodents also modify their environments (Brown and Harney 1993; Price and Jenkins 1986). They dig burrows, which moves the soils and provides habitat and refugia for other species, including other rodents, reptiles, amphibians, birds and invertebrates. Collection, storage and consumption of seeds by kangaroo rats, for example, has profound effects on the vegetation structure of the habitats they occupy (Price and Jenkins 1986). In addition, resource use by pocket mice and kangaroo rats substantially overlaps with that of seed-eating birds and harvester ants. However, in a literature review of effect of granivorous rodents on the plant community, Price and Jenkins (1986) cautioned against drawing broad generalizations because specific effects will be affected by competitor densities, climate and edaphic conditions, rodent densities, seed preferences, and caching behavior.

The coevolutionary results of such inter- and intraspecific community relationships and their relationship to plant communities are not understood, but it can be concluded that rodents are an important component of arid ecosystems. In addition to their direct impacts on plant communities, they are important prey for a variety of predators and their presence also affects populations of other prey such as small reptiles, lagomorphs and some birds (Brown and Harney 1993).

Physiological Ecology: The little pocket mouse has demonstrated several physiological adaptations that allow it to survive in extreme and unpredictable environments. Perhaps best known is its ability to enter torpor or hibernate for long periods during the cold winter months. This trait is thought to be a means to conserve hoarded food during their seasonal dormancy and reflects the cost-benefit ratio of foraging on the surface during the winter (Kenagy 1973). Little pocket mice enter torpor through slow-wave sleep, which may itself be a mechanism for energy conservation in

many species (French 1993). The timing of torpor and dormancy appears to be at least partly endogenously controlled because little pocket mice show distinct phases of dormancy and activity under constant conditions of temperature, photoperiod and food availability in the laboratory (French 1993). Also, the disappearance of mice from the surface in the wild is asynchronous (O'Farrell 1974) and the cycle of dormancy can be changed by hormonal manipulation and not allowing animals to build up a food hoard (French 1993). Kenagy (1973) reported that little pocket mice can remain torpid for more than 72 hours at 3% of their normal basal metabolic rate (BMR).

Another physiological mechanism that allows little pocket mice to survive in extreme environments is a low BMR. Their BMR is 51-81% of that expected based on their body mass. Also, this species has been demonstrated to rest at their lower end of thermoneutrality whenever possible (French 1993).

Little pocket mice have relatively low rates of evaporative water loss compared to most mammals that is accomplished through a reduction in respiratory and cutaneous water losses (French 1993). It is not known whether little pocket mice are completely independent of exogenous water, as are at least three other heteromyids (*Dipodomys merriami*, *Chaetodipus fallax*, and *C. penicillatus*). Other potential mechanisms for conserving water include reduced fecal water loss and reduced lactational water loss.

Potential behavioral adaptations for maintaining water balance, energy, and thermoneutrality are remaining in day burrows during periods of climatic extremes, plugging burrow entrances to retain moisture (*i.e.*, humidity) in the burrow (Kenagy 1973), and ingestion of herbaceous and succulents plants (possibly to support lactation). Kenagy also found that little pocket mice in the Great Basin Desert position themselves in their burrow in relation to soil temperatures that vary daily and seasonally. For example, in the early spring, little pocket mice moved from a depth of 30-40 cm where the temperatures were 12 to 14 degrees Celsius to within 1 cm of the surface by midmorning, where temperatures reached 29 degrees Celsius by midday.

Known Distribution

The historic range of the Los Angeles pocket mouse was estimated to be from Burbank and San Fernando in Los Angeles County east to the City of San Bernardino, San Bernardino County (the type locality) (Hall 1981). Its range extends eastward to the vicinity of the San Geronio Pass in Riverside County, and southeast to Hemet and Aguanga, and possibly to Oak Grove, in north-central San Diego County (Hall 1981; Patten *et al.* 1992).

Status in Wash Planning Area

There are only 7 documented occurrences of the Los Angeles pocket mouse in the Wash Planning Area, all of which are from studies by the San Bernardino County Museum and all concentrated in the eastern half of the Wash Planning Area. Three of the occurrences are in intermediate alluvial fan sage scrub, one in intermediate-mature alluvial fan sage scrub, and two are in pioneer alluvial fan sage scrub. These six occurrences are all associated existing floodplain and water way uses. One location is mapped in developed habitat within the existing water conservation area. Although there are only seven documented occurrences, it is assumed that the Los Angeles pocket mouse could occur throughout the Wash Planning Area in alluvial fan sage scrub, with the largest populations occurring in intermediate alluvial fan sage scrub.

Special Biological Considerations

One of the most unique aspects of little pocket mouse biology, and one that makes it difficult to study, is its trait of entering long periods of dormancy during the winter. Some have suggested (*e.g.*, Chew and Butterworth 1964) that this dormancy is related to its longevity (although it was noted above that year-to-year survival was positively related to winter surface activity, mild weather conditions and high food production [Kenagy 1973]). Also, this species may not breed during poor conditions (O'Farrell 1974) and, as a result, may limit surface activity some years. These traits make this species difficult to census and monitor population trends.

Threats to Species

Habitat Loss and Fragmentation: Urbanization, agriculture, sand and gravel mining, and flood control projects are serious threats to the Los Angeles pocket mouse. Loss of and disruptions in the continuity of drainages and alluvial fan habitats that support patchy distributions of the species probably results in isolation of local populations and preclude or limit the amount of genetic exchange between populations. Such isolation can result in genetic drift and loss of heterogeneity in the populations, leaving small local populations at high risk of extirpation. Furthermore, the loss of large areas of sandy loam habitats in occupied bottom lands may also adversely affect this subspecies (S. Montgomery 1998).

Disease: Whitaker *et al.* (1993) report a variety endo- and ectoparasites and associates carried by the little pocket mouse. Little pocket mice carry rickettsia, which are small, non-motile or bacterial-like organisms, including *Coxiella burnetii* that causes Q fever and *Rickettsia rickettsii* which causes Rocky Mountain spotted fever. Both are carried by tick vectors. One flagellate protozoan, *Tritrichomonas muris*, is carried by the little pocket mouse. One tapeworm (Cestoda), *Mathevotaenia deserti*, and one roundworm (Nematoda), *Protospirura dipodomis*, also have been reported in little pocket mice. Mites (excluding chiggers) found on little pocket mice include

Androlaelaps fahrenheitsi, *Echinonyssus hilli*, *E. incomptis*, *E. triacanthus*, *E. utahensis*, *Eubrachyla elaps circularis*, *Hypoaspis leviculus*, *Ischyropoda armatus*, *I. furmani*, and *Sertitympanum* sp. Chiggers found on little pocket mice include *Dermadelema furmani*, *D. lynnae*, *D. mojavnense*, *D. sleeperi*, *Euschoengastia decipiens*, *E. heteromyicola*, *E. obscura*, *E. stephensi*, *Euschoengastoides imperfectus*, *Eutrombicula belkini*, *Hexidionis deserti*, *H. doremi*, *Hyponeocula arenicola*, *H. fovea*, *H. imitator*, *Odontacarus linsdalei*, *Otorhinophila desertorum*, and *O. xerophila*. Ticks reported from little pocket mice include *Dermacentor parumapterus*, *Ixodes kingi*, and *I. sculptus*. Finally, fleas reported from little pocket mice include *Meringis dipodomys*, *M. hubbardi*, *M. parkeri*, and *Rhadinopsylla sectilis*. It is not known how harmful these parasites and associates are to little pocket mice, or what level of mutualism has evolved (e.g., benefits that might occur to the host) (Whitaker *et al.* 1993).

3.2.11 Speckled Dace

SPECIES DESCRIPTION

Status

The speckled dace is not state- or federally-listed but the Santa Ana subspecies is a California Special Concern species and has been identified only as subspecies 3 due to difficulties in determining where the subspecies is valid. The species is also considered a USDA Forest Service sensitive species. For the purposes of this analysis, the Santa Ana subspecies of the speckled dace is considered to be valid (see discussion below under "Genetics").

Habitat and Habitat Associations

Santa Ana subspecies of speckled dace occurs in permanent flowing streams with water temperatures between 17 and 20 degrees C (63 to 68 degrees F) that flow from cool springs. Typical stream habitat, such as the West Fork of the San Gabriel River, is fairly wide (15 to 25 feet) and shallow (4 to 8 inches), with typical summer flow of four cubic feet per second. Dace are most commonly found in the lower reaches of streams consisting primarily of runs and riffles with gravel and cobble substrates. (Deinstadt, *et al.* 1990).

Biology

Genetics: Speckled dace, one of the most widely distributed freshwater fish in the western United States, is a small (less than 9 mm TL) member of the Cyprinidae (minnow) family. Although not formally described as a subspecies, much data exists warranting this status. In 1996, the U.S. Fish and Wildlife Service determined that there

was insufficient information provided to substantiate that the Santa Ana subspecies of speckled dace is a described subspecies (USFWS 1996) and denied a petition to list it. The primary support for subspecific status includes reference to a Master's thesis (Cornelius 1969) which was not included with the petition and reference to unpublished genetic data. Other, anecdotal evidence supporting subspecific status includes a species account written by C.C. Swift that was included as pages 207-212 in a document entitled Fishes, Aquatic Diversity Management Areas, and Endangered Species: A Plan to Protect California's native Aquatic Biota, edited by Moyle and Yoshiyama (1992).

Diet and Foraging: Speckled dace generally feed on small invertebrates in stream bottoms (Moyle 1976). Diet preference of speckled dace in the Trinity River varied seasonally, with chironomid larvae preferred in winter, mayfly and stonefly nymphs in the spring, flying insects in the summer, and filamentous algae in the fall (Jhingran 1948). Other food items of speckled dace are eggs and larvae of suckers and other minnows (Moyle *et al.* 1995), ephemeropteran nymphs, and ostracods (Angradi *et al.* 1991).

Daily Activity: Although dace are not typically solitary, they avoid obvious schooling behavior except during breeding season (June and July). Speckled dace in the Trinity River spend the day resting/hiding among rocks or in slightly deeper water, and are most active at night. In general, speckled dace are active year-round (Moyle 1976).

Reproduction: No specific information on reproduction of the Santa Ana subspecies of speckled dace are available but life history characteristics are presumably similar to other stream dwelling speckled dace (Moyle *et al.* 1995). Speckled dace generally become mature in their second summer, with most spawning occurring in June and July, induced by rising water temperatures. Speckled dace spawn on the gravel edges of riffles, with the males creating a bare patch of rocks and gravel for spawning by removing algae and detritus. A female entering this area is surrounded by a knot of males, who will release sperm simultaneously with the female's release of a few eggs (John 1963). Eggs hatch in six days and larval fish remain in the substrate for up to eight days.

Survival: Life history data for the Santa Ana subspecies of speckled dace indicate that it probably lives for three years (Deinstadt *et al.* 1990). Specific survival characteristics are presumably similar to other stream dwelling speckled dace (Moyle *et al.* 1995). In a comparison of five fish species in the upper Klamath Basin in Oregon, speckled dace were the second most susceptible to low dissolved oxygen levels, an indication that eutrophication may adversely affect this species (Castleberry and Cech 1992). Extreme thermal variance, competition for food and space with other native and nonnative fishes, and predation by other fishes provide challenges to its survival. Conversely, a highly

developed sense of hearing and smell, pharyngeal teeth, and a high fecundity rate aid in its survival.

Dispersal: Although the Santa Ana subspecies of speckled dace was considered to have been extirpated from the Los Angeles River drainage by Moyle *et al.* 1995), after a thorough search of Big Tujunga Creek in 1990 - 1992, ten speckled dace of the Santa Ana subspecies were again collected in Big Tujunga Wash and part of Haines Creek in 2002 (CNDDDB 2004). Speckled dace in high desert streams in Oregon were exceptionally good at recolonizing stream reaches after flash floods removed fishes from specific reaches (Pearsons *et al.* 1992).

Community Relationships: The Santa Ana subspecies of speckled dace is most common where other native fishes, such as rainbow trout and Santa Ana sucker are common and low numbers of introduced species are present. Brown trout are believed to prey on the Santa Ana subspecies of speckled dace (Deinstadt *et al.* 1990).

Known Distribution

The Santa Ana subspecies of speckled dace is considered to be limited to headwaters of the Santa Ana, San Gabriel, and Los Angeles Rivers (CNDDDB 2004). In the San Gabriel River system, it has been reported from the East, North, and West forks, Big Mermaids Canyon Creek, and Bear Creek (CNDDDB 2004). Although Moyle *et al.* (1995) concluded that speckled dace had been extirpated from the Los Angeles River system, it was reported from Big Tujunga Creek and part of Haines Creek in 2002 (CNDDDB 2004). In the Santa Ana River system, it is reported from the Santa Ana River downstream of Van Buren Street (Riverside County), Santiago Creek (Orange County), Cajon Creek, north fork of Lytle Creek, Strawberry Creek, Shrewsberry Springs in Silverado Canyon, Plunge Creek, and Mill Creek (San Bernardino County) (CNDDDB 2004, Moyle *et al.* 1995). The latter two streams are within the project area.

Status in Wash Planning Area

Data reviewed includes the CNDDDB, the USFWS database, the San Bernardino County Natural History Museum, the UCR herbarium and available literature. There is one occurrence of speckled dace recorded in the Wash Planning Area; this occurrence is from the San Bernardino County Natural History Museum database and was observed between 1997 and 2000. The occurrence was mapped along the Santa Ana River, just west of Greenspot Road. The vegetation associated with the speckled dace location includes pioneer Riversidean alluvial fan sage scrub.

Special Biological Considerations

Members of the Cyprinidae family are especially sensitive to acoustic disturbances due to an acute sense of hearing. Cyprinids have a set of specialized bones (Weberian ossicles) connecting the air bladder to the inner ear, allowing sound waves to travel easily as vibrations are sensed underwater (Moyle 1976). Therefore the speckled dace may be susceptible to adverse impacts from construction in riparian areas due to the associated noise.

Threats To Species

The Santa Ana subspecies of speckled dace appears to have been extirpated from the San Jacinto River, and undergone marked population reduction in several other streams due to loss of habitat, population isolation due to dam construction, erratic water flows from dams and other diversions, introduction of nonnative species, recreational uses that alter stream habitats and disturb reproduction, degradation of water quality, and low water levels due to drought (Moyle, *et al.* 1995). Recommended management measures to protect remaining populations include securing adequate water for fish survival and reintroduction into drainages where populations have been extirpated (Moyle, *et al.* 1995).

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