For each plant or animal addressed by the Plan, a ***Species Account*** was prepared. The Supergroup approved the list of 98 plant and animal species to be addressed by the Plan in 1996. The USGS then contracted with experts on each species, who prepared the species accounts for use in development of the Plan. A wildlife biologist or botanist possessing recognized expertise concerning the species in question authored each of these documents. These accounts describe the general status, habitat, life history, distribution, biological goals, and threats faced by each species, as well as a detailed bibliography. All species accounts were peer reviewed.

Desert Tortoise
Gopherus agassizii

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Legal Status: Federal - Threatened
State - Threatened

General Distribution: The desert tortoise is widely distributed throughout major portions of the Mojave and Sonoran deserts of California, Nevada, Utah, Arizona, Sonora, and Sinaloa. Genetic, morphological, ecological, and behavioral features suggest an evolutionary divergence between the tortoises found south and east of the Colorado River ("Sonoran population"), and those found north and west of the river ("Mojave population;" Lamb et al. 1989). The latter is the population Federally and State-listed as threatened. This population will be referred to in the remainder of this account. The majority of animals in the Mojave population occur at variable densities in six distinct population segments (i.e., evolutionarily significant units), each identified in the Recovery Plan for desert tortoises as separate Recovery Units (USFWS 1994).

Distribution in the WMPA: One major segment of the Mojave population of the desert tortoise occurs almost entirely within the WMPA and is called the West Mojave Recovery Unit (USFWS 1994). Tortoises in the West Mojave Desert are divided into four not entirely separate subpopulations: Ord-Rodman, Superior-Cronese, Fremont-Kramer, Joshua Tree (see USFWS 1994 for detailed descriptions). Population densities within these areas are variable and patchy varying from 0-250/mi^2 (0-155/km^2; Berry and Nicholson 1984). Within each subsegment, tortoise density is highly heterogeneous with clusters of high densities (perhaps several hundred hectares in size) surrounded by areas of rather low densities in seemingly suitable habitat. Tortoises occur outside of these four subsegments, but at very low densities (i.e., 0-20/mi^2 (0–12/km^2; Berry and Nicholson 1984).

Natural History: The desert tortoise is a medium-sized, terrestrial turtle in the family Testudinidae. The shell is light brown to very dark brown with brown to orange or yellow in the centers of scutes, particularly in young animals. The skin is dry and scaly with thick, stumpy,

elephantine hind legs. A strong projection, the gular horn, located at the anterior end of the plastron, is most pronounced in adult males. Adult males also have shorter claws, longer, thicker tails, a concave plastron, and pronounced chin glands. They weigh 0.04-10+ lbs (20-5000+ g) and range in size from about 1.4 inches (35 mm; carapace length) at hatching to 11-16 inches (280-400 mm; carapace length) as adults. No other terrestrial turtle occurs within the range of the desert tortoise.

Desert tortoises are long lived with delayed sexual maturity. Some individuals begin reproducing when 7.4 inches (180 mm) long (median carapace length, MCL), which they attain when about 12-15 years old. The majority do not begin reproducing until they reach 8.2 inches (208 mm; approximately 12-20 years old; Turner and Berry 1984, Turner et al. 1986). Maximum longevity in the wild is likely to be about 50 to 70 years, the norm being 25 to 35 years (Germano 1992, 1994). The average clutch size is 4.5 eggs (range 1-8), with 0-3 clutches deposited per year (Turner et al. 1986). Clutch size and number probably depend on female size, water, and annual productivity of forage plants in the current and previous year (Turner et al. 1984, 1986; Henen 1997). The ability to alter reproductive output in response to resource availability may allow individuals more options to ensure higher lifetime reproductive success. The interaction of longevity, late maturation, and relatively low annual reproductive output causes tortoise populations to recover slowly from natural or anthropogenic decreases in density. To ensure population stability or increase, these factors also require relatively high juvenile survivorship (75-98% per year), particularly when adult mortality is elevated (Congdon et al. 1993).

Most eggs are laid in spring (Apr -Jun) and occasionally in fall (Sept-Oct). Eggs are laid in sandy or friable soil, often at the mouths of burrows. Hatching occurs 90-120 days later, mostly in late summer and fall (mid Aug-Oct). Eggs and young are untended by the parents. Tortoise sex determination is environmentally controlled during incubation (Spotila et al. 1994). Hatchlings develop into females when the incubation (i.e., soil) temperature is greater than 89.3° F (31.8° C) and males when the temperature is below that (Spotila et al. 1994). Mortality is higher when incubation temperatures are greater than 95.5° F (35.3° C) or less than 78.8° F (26.0° C). The sensitivity of embryonic tortoises to incubation temperature may make populations vulnerable to unusual changes in soil temperature (e.g., from changes in vegetation cover), but there are no data available from the field that can be used to test this hypothesis.

Tortoise activity patterns are primarily controlled by ambient temperature and precipitation (Nagy and Medica 1986, Zimmerman et al. 1994). In the East Mojave and

Colorado Deserts, annual precipitation occurs in both summer and winter, providing food and water to tortoises throughout much of the summer and fall. Most precipitation occurs in winter in the West Mojave Desert resulting in an abundance of annual spring vegetation, which dries up by late May or June. Tortoises in this region are primarily active between May and June, with a secondary activity period from September through October. Tortoises may also be active during periods of mild or rainy weather in summer and winter. During inactive periods, tortoises hibernate, aestivate, or rest in subterranean burrows or caliche caves, and spend approximately 98% of the time in these cover sites (Marlow 1979, Nagy and Medica 1986). During active periods, they usually spend nights and the hotter part of the day in their burrow; they may also rest under shrubs or in shallow burrows (called pallets). Tortoises use an average of 7-12 burrows at any given time (Barrett 1990, Bulova 1994, TRW Environmental Safety Systems Inc. 1997); some burrows may be used for relatively short periods of time and then are replaced by other burrows. Tortoises sometimes share a burrow with several other tortoises (Bulova 1994).

Tortoises eat primarily annual forbs, but also perennials (e.g., cacti and grasses). Forage species selected by tortoises in the west Mojave Desert include: *Astragalus didymocarpus*. *Astragalus layneae*, *Camissonia boothii*, *Euphorbia albomarginatus*, *Lotus humistratus*, and *Mirabilis bigelovii* (Jennings 1993). In the east Mojave Desert, tortoises showed a preference for *Camissonia boothii*, *Cryptantha angustifolia*, *Malacothrix glabrata*, *Opuntia basilaris*, *Rafinesquia neomexicana*, *Schismus barbata*, *Stephanomeria exigua* and other species (Avery 1998). On rare occasions they have been observed eating other items such as caterpillars, lizards, and cow dung, but these make up a very small proportion of their diets (Jennings 1993, Esque 1994, Avery 1998). Although they will eat exotic plants, tortoises generally prefer native forbs when available (Jennings 1993, Avery 1998, cf. Esque 1994). The dietary preference may place them at a nitrogen and water deficit. Droughts frequently occur in the desert, resulting in extended periods of low water availability. Periods of extended drought place tortoises at even greater water and nitrogen deficit than during moderate or high rainfall years (Peterson 1996, Henen 1997). During a drought, more nitrogen than normal is required to excrete nitrogenous wastes, thus more rapidly depleting nitrogen stored in body tissues. Plants also play important roles in stabilizing soil and providing cover for protection from predators and heat.

The tortoise mating system is probably polygynous, and may be polyandrous, although DNA fingerprinting to analyze patterns of paternity has not been conducted. Choice of mate is mediated by aggressive male-male interactions and possibly by female choice (Niblick et al. 1994). Recent findings indicate that tortoises in the West Mojave Desert may exhibit pre-breeding dispersal movements, typical of other vertebrates, ranging from 1 to 10 miles (0.6-16

km) away in a single season (Sazaki et al. 1995). The advantage of pre-breeding dispersal may be to find a more favorable environment (physical, biotic, social) in which to reproduce. However, the risk is increased mortality from predation, exposure, starvation, or anthropogenic factors (e.g., motor vehicle mortality).

Tortoise activities are concentrated in core areas, known as home ranges. These home ranges overlap; because tortoises do not defend a specific, exclusive area, they do not maintain territories. Home range sizes have been measured at 10-450 acres (4-180 hectares) and vary with sex, age, season, and density or availability of resources (USFWS 1994). Whereas home range sizes may vary from year to year, it is not known at what rate tortoises change their home range location and size over the course of their life. Over their entire life span, an individual tortoise may require considerably larger areas than that used in individual years.

There are many natural causes of mortality, but their extents are difficult to evaluate and vary from location to location. Several native predators are known to prey on tortoise eggs, hatchlings, juveniles, and adults including: coyote (*Canis latrans*), kit fox (*Vulpes macrotis*), badger (*Taxidea taxus*), skunks (*Spilogale putorius*), common ravens (*Corvus corax*), golden eagles (*Aquila chrysaetos*), and Gila monsters (*Heloderma suspectum*). Additional natural sources of mortality to eggs, juvenile, and adults may include desiccation, starvation, being crushed (including in burrows), internal parasites, disease, and being turned over onto their backs during fights or courtship (Luckenbach 1982, Turner et al. 1987, pers. obs.). There are little data available to evaluate the relative contributions any of these factors make to natural mortality in undisturbed tortoise populations. Population models indicate that for a stable population to maintain its stability, on average, no more than 25% of the juveniles and 2% of the adults can die each year (Congdon et al. 1993, USFWS 1994). However, adult mortality at one site in the West Mojave was 90% over a 13-year period (Berry 1997). Morafka et al. (1997) reported 32% mortality over five years among free-ranging and semi-captive hatchling and juvenile tortoises (up to 5 years old) in the West Mojave. When the 26 that were known to have been preyed on by ravens were removed from the analysis, mortality dropped to 24%. Turner et al. (1987) reported an average annual mortality rate of 19 - 22% among juveniles over a nine year period in the East Mojave.

Habitat Requirements: Vegetation and topography in tortoise habitat within the WMPA are variable. The greatest population densities in the WMPA are found in creosote bush scrub with lower densities occurring in Joshua tree woodland and Mojave-saltbush-allscale scrub. Major topographical features used by tortoises include flats, valleys, bajadas, and rolling hills generally

from 2000-3300 ft (600-1000 m) in elevation and occasionally above 4100 ft (1250 m; Weinstein 1989). Tortoises typically avoid plateaus, playas, sand dunes, steep slopes (>20%) and areas with many obstacles to free movement. They prefer surfaces covered with sand and fine gravel versus course gravel, pebbles, and desert pavement (Weinstein 1989). Friable soil is important for digging burrows, but when friability (e.g., diggability) is similar, productivity of plants is more important (Wilson and Stager 1992).

In an attempt to quantify the relationship between tortoise abundance and habitat characteristics, Weinstein (Weinstein et al. 1987, Weinstein 1989) found habitat to be difficult and complex to characterize with any accuracy. Food availability, soil diggability, longitude (higher densities in West Mojave Desert), and degree of stream-washing were the habitat characteristics that were most useful in discriminating between areas with high densities of tortoises and those with no tortoises. However, the model was quite poor at classifying into correct density categories data that were not used in developing the model.

Population Status: It is commonly claimed that tortoise populations have suffered drastic declines throughout much of the species' range, but a thorough presentation of these data has never been published (Bury and Corn 1995). Nonetheless, the cursory published accounts of tortoise populations in the West Mojave Desert do show significant reductions, at least in that region (Corn 1994, Berry and Medica 1995). At one site in the Desert Tortoise Natural Area, north of California City, a loss of approximately 76% was estimated to occur between 1979 and 1992 followed by no apparent change in 1996 (Berry 1997, Brown et al. 1999). A complete analysis of the existing data is needed. Most of the deaths were thought to be caused by a respiratory disease (see below).

Threats Analysis: Direct threats to desert tortoise populations are those that immediately affect survival and reproduction and are much easier to document. Indirect threats are those that may affect individuals in some less immediate way, such as by reducing food or altering the soil temperature, which then may affect tortoise reproduction or survival. Indirect threats are often very difficult to substantiate. Examples of direct threats include: collisions with motorized vehicles, illegal collecting, and disease. Indirect threats likely affecting tortoise populations include: habitat loss from construction and agricultural development; habitat alterations from livestock grazing, recreational activities, atmospheric pollution, global warming, and invasions of exotic plants.

Two general phenomena are particularly critical to understanding trends in tortoise populations. Their life history characteristics (e.g., delayed maturation, longevity, low average annual reproductive output, and highly variable nest success) make them susceptible to increased mortality and long delays in recovering from catastrophic losses. Additionally, as populations become increasingly fragmented, the probability of population persistence becomes more tenuous. When population size is low, inbreeding becomes a potential problem. Smaller populations also are at an increased risk of extinction from catastrophes (e.g., fires and disease) and random variation in population parameters, like sex ratio, age class structure, fecundity, and mortality. Populations can reach non-recoverable levels through fragmentation into smaller populations (e.g., from highways, utility corridors, and development) and exacerbated mortality within these fragmented populations.

The relative importance of different threat factors is difficult to rate. First, the cause of death of animals and how much decline is really attributable to the various indirect causes of mortality (e.g., habitat alteration) is difficult to determine. Second, too little is known about several potential threats to evaluate their absolute or relative impacts. Third, determining which factors cause mortality is very site specific. The following evaluation of the relative importance of each factor in terms of threat to tortoise population viability in the West Mojave Desert is based on the available scientific evidence, which is often incomplete. More complete discussions can be found in Lovich and Bainbridge (1999) and Boarman (2002).

The greatest threats to tortoise populations in the WMPA are probably: disease; the cummulative effects of habitat loss, degradation, and fragmentation from construction, urbanization, and development; and a high level of human access to tortoise habitat. Disease, specifically Upper Respiratory Tract Disease (URTD), may have caused dramatic declines in some populations (Berry 1997). The causative agent of URTD is the bacterium *Mycoplasma agassizii*, which causes lesions in the respiratory tract (Jacobson 1994). Clinical signs of the disease (e.g., swollen eye lids, nasal discharge, wheezy breath, and in extreme cases, lethargy) have been found in many animals within populations experiencing high mortality rates and has been found in some animals prior to death. The introduction, or at least the spread of the disease in some populations, has been attributed to the release of infected captive animals. Little is known about the epidemiology of the disease. It is also unclear if the disease is actually lethal and some apparent recovery has been observed (Brown et al. 1994a,b). Other than preventing the spread by proper handling of animals by trained workers, nothing is known about how to reduce the effects of the disease in wild populations. A shell disease, cutaneous dyskeratosis, has also been identified within tortoise populations. Cutaneous dyskeratosis has been associated

with high mortalities in the Chuckwalla Bench area of the Colorado Desert and may be caused by a vitamin or mineral deficiency or contact with a natural or anthropogenic toxicant in the environment (Jacobson et al. 1994, Homer et al. 1998). Little is known about the cause, epidemiology, or treatment of this shell disease, and its incidence appears to be low in the West Mojave Desert.

Habitat loss, degradation, and fragmentation are major problems in the WMPA because of the high level of human activity. The West Mojave Desert is a growing suburban area, with an increase in housing, industry, and commercial development in major municipal areas, rural areas, and along major transportation corridors. The loss of habitat, mortality from increased traffic, reduced quality of habitat altered by human presence and activity, fragmentation of populations, and the cumulative effects of other problems associated with humans (e.g., dogs, recreation, utility corridors, etc.) pose a significant and increasing problem for the viability of tortoise populations within the WMPA.

Many of the individual threats discussed below relate to the level of access to tortoise habitat afforded to people. For instance, illegal collecting of tortoises for food or cultural ceremonies has been documented on a few occasions by law enforcement officials (USFWS 1994). There is ample evidence that driving off of roads compacts soil and damages vegetation (see "ORV" section, below). The possibility also exists that tortoises or their burrows may be crushed. Even though off-road rehicle (ORV) activity on roads may pose little such direct impact to tortoises or their habitat, the presence of a road poses potential harm to tortoises and their habitat, and the more roads there are the greater is the proportion of the tortoise population that is under the threat of harmful off-road activity. Other potentially harmful activities that likely occur in greater numbers near roads include: mineral exploration, illegal dumping of garbage and toxic wastes, release of ill tortoises, anthropogenic fire, handling and harassing of tortoises, spread of invasive weeds, and trailing of sheep (Berry and Nicholson 1984). The threat posed to tortoise populations by each of these activities likely increases with increased access afforded by the proliferation of roads, even very lightly traveled ones. Furthermore, some of these individual threats may be relatively low, but their cumulative impact may be great.

Several activities may be considered "moderate threats" to tortoise populations in the WMPA because they cause less direct mortality, are less widespread, are not likely to increase, probably pose a relatively low to moderate level of risk to tortoises or tortoise populations, or little is known about their impacts. The importance of each threat varies from place to place.

Also, individually, these threats may be of moderate importance, but their overall cumulative effects are probably of extreme importance.

Agriculture. The effect of agriculture on tortoise populations is primarily through the loss of habitat. When tortoise habitat is converted to agricultural use, it becomes largely unsuitable to tortoises. Other impacts include the introduction of invasive weeds, facilitation of increases in raven population, lowering of the water table, production of dust, and possible introduction of toxic chemicals.

Fire. Fire is an ever increasing threat to tortoises and their habitat in the WMPA. Fire was previously rare in the Mojave Desert, but has increased with the proliferation of introduced plants, particularly the grass red brome (*Bromus rubens*), which provide fuel for fires. Red brome helps to spread fire because it is common, tends to grow in large relatively dense mats, and fills the intershrub spaces, which are largely devoid of much native vegetation (Brooks 1998). Fires can cause direct mortality when tortoises are burned, which can happen both inside and out of burrows. There are a few documented examples of tortoises being burned by fires (Homer et al. 1988, Esque et al. in press). Other indirect impacts fires may have on tortoise populations include: 1) short-term effect of removing dry and some living forage plants (but this effect is likely short-term); 2) long-term effects of facilitating proliferation of non-native plants, which are of lower nutrient value to tortoises (Avery 1995), and may be avoided by them; 3) short-term fragmentation of tortoise habitat by creating patches of unsuitable habitat, at least on the short term; 4) alteration of temperature profile from removal shade; 5) loss of shrubs used as daytime or night time cover sites; and 6) decreased soil stability and increased erosion.

Landfills. In the West Mojave Desert, there are 13 county-run solid-waste landfills and an unknown number of unauthorized dumpsites. The potential impacts of landfills on tortoise populations include: loss of habitat, spread of garbage, introduction of toxic chemicals, increased road mortality, and proliferation of predatory species. The loss of habitat from landfill presence and expansion is relatively minor except when viewed in the context of habitat degradation and fragmentation caused by the myriad human developments that are proliferating in the desert. The greatest potential impact from landfills is their probable role in facilitating the increase in populations of predators such as common ravens, and perhaps coyotes. Ravens make extensive use of landfills for food (Boarman 1993). The food eaten probably supports raven populations through the summer and winter, when natural resources are in low abundance. As a result, large numbers of ravens are present at the beginning of the breeding season (Feb-Jun). Some then move into tortoise habitat, and then nest, raise young, and potentially feed on

tortoises. Predation by ravens is probably relatively low within the immediate vicinity of landfills because of the low density of tortoises in the vicinity, but predation may increase as ravens disperse to nest farther from the landfills (Kristan 2001).

Grazing. Grazing by cattle and sheep has several potential direct and indirect effects on tortoise populations. These include: mortality from crushing of animals or their burrows, destruction of vegetation, alteration of soil, augmentation of forage (e.g., presence of livestock droppings, and stimulation of vegetative growth or nutritive value of forage plants), and competition for food. There is weak evidence for declines in tortoise density directly associated with grazing, but its evaluation is complicated by the presence of multiple factors affecting tortoises at most sites and the difficulty of being able to measure accurately tortoise densities to assay direct effects (Luke et al. 1991, Oldemeyer 1994). There are observations of sheep or cattle stepping on tortoises or their burrows (Berry 1978, Nicholson and Humphreys 1981, Avery 1998). Cattle may out compete tortoises for some seasonally important forage species (i.e., desert dandelions, *Malacothrix glabrata;* Avery 1998), but the few studies testing for it do not show strong effects of competition (Tracy 1996). Past studies have shown dietary overlap, a condition necessary, but not sufficient, to show competition (Avery 1998). There are only two studies showing sheep and tortoises eat some of the same food items (Hansen et al. 1976, Nicholson and Humphreys 1981). However, there are no studies that tested if sheep compete with tortoises for food. If livestock significantly affect tortoise populations, it is most likely through habitat alteration. Sheep and cattle are known to compact soil, trample vegetation, and cause observable changes in the composition and structure of the plant and animal communities (Nicholson and Humphreys 1981, Webb and Stielstra 1979, Berry 1978, Brooks 1995, Avery et al. in prep.). No evidence is available to indicate that sheep or cattle benefit tortoises by providing food or improving habitat condition in the Mojave Desert (cf. Bostick 1990).

Military. There are five military bases located within the WMPA. Impacts of military activities on tortoises vary from base to base, but generally fall into four categories. Four of the five facilities have large internal support communities, while all five have large operations areas. Both of these factors result in loss, degradation, and fragmentation of habitat. Each one of the bases also support local communities (e.g., Barstow, Ridgecrest, Twenty-nine Palms), which also destroy, degrade, and fragment habitat. Four of the five bases conduct substantial field maneuvers (e.g., tank operations, detonation of air and ground based explosives) that can result in direct mortality for tortoises and destruction and degradation of tortoise habitat. The proposed expansion of Fort Irwin southward will degrade a portion of tortoise habitat used by the Superior-Cronese population; this proposed expansion may pose the greatest single threat to the

West Mojave tortoise population. An additional, but unexplored, possible impact of military activity is contamination by toxic chemicals.

By virtue of the restrictive, secret, or hazardous nature of much of their activities, military bases sometimes offer a great level of protection to tortoise populations. For example, restrictions on access to the Precision Impact Range Area at Edwards Air Force Base reduces the amount of human traffic in the area, which is designated critical habitat. All bases, to some extent, prohibit public access, which likely results in less ORV activity, shooting, dumping, etc., than on adjacent private lands. The 44 square miles (71 km^2) of NASA's Goldstone Deep Space installation on Fort Irwin is a good example of relatively protected habitat where dog tracks, shotgun shells, sheep scat, etc. are scarce (LaRue pers. comm.). Furthermore, each of the bases is required to develop a Natural Resource Plan, which must comply with the Endangered Species Act.

Off-road Vehicles. Off-road vehicle (ORV) activities have caused a substantial loss of tortoise habitat and a reduction in its quality. Competitive events and free play activities cause destruction and degradation of vegetation, compaction of soil, a reduction in tortoise densities, and likely crush tortoises and burrows (Davidson and Fox 1974, Vollmer et al. 1976, Adams et al. 1982, Webb 1983, Bury and Luckenback 1986, Berry 1990 as amended). Some designated Open (free-play) Areas are in formerly high tortoise density areas, so current ORV activities may prevent recolonization of former tortoise habitat. Little data are available to evaluate the effects of light OHV activity on tortoise populations or their habitat, but the direct effects are likely to be minor if vehicles use designated routes of travel and stay on the roads. Indirect impacts, which may be substantial, probably occur wherever vehicle access is allowed in tortoise habitat. These impacts potentially include: soil compaction, vegetation destruction, significant disturbance of biotic soil crusts (i.e. cryptogams), increased soil destabilization and erosion, proliferation of non-native weeds, crushing of tortoise and burrows when vehicle leave the road, shooting and vandalism of tortoises, harming of tortoises or their burrows by dogs, deposition of garbage, providing food for ravens, and the handling, collecting, or disturbance of tortoises. Data necessary to evaluate the extent of these effects are not available, although each of these impacts is to some degree more prevalent in areas with roads than in roadless areas. Berry et al. (1994) found a negative correlation between off-road vehicle trails and tortoise sign (an index of tortoise density), although that may be because tortoise sign were obliterated by vehicles or covered by dust.

Predation. Predation is a naturally-occurring phenomenon. Predation by common ravens has become a major problem for some tortoise populations. Raven populations are increasing at a precipitous rate because of resource subsidies (food, water, nesting substrate) that are provided by increasing human populations (Boarman 1993). Ravens prey on juvenile tortoise (mean size = 2.7 inches [68.4 mm; range = 1.3-4.9 inches; 33-124 mm]). The remains of juveniles killed by ravens have been found throughout the WMPA (Berry 1985, Boarman unpubl. data). Between 1968 and 1992, raven populations in the Mojave Desert have increased by over 1000% (Boarman and Berry 1995), with the highest increases probably occurring in the West Mojave Desert. This means that every year there are more ravens present in tortoise habitat, thus increasing the predation pressure on tortoise populations. Predation by domestic dogs (*Canis familiaris*) and coyotes are likely depleting some tortoise populations (Berry 1990, as amended, Bjurlin and Bissonette 2001). Few data are available to evaluate the nature of the problem caused by these two species, which also benefit from human-based resource subsidies.

Road Mortality. Roads and highways have several impacts on desert tortoise populations and their habitat (Boarman and Sazaki 1996). Direct impacts include mortality through road kills and destruction of habitat (including burrows). On a series of annual surveys, Boarman and Sazaki (1996) found an average of one dead tortoise (mostly adults and subadults) per year for every two miles of highway. This was a conservative estimate of the incidence of road kill along California State Highway 395. Indirect effects include degradation of habitat because the roads serve as corridors for dispersal of invasive weeds, predators, development, recreation, and other anthropogenic sources of impact. They also fragment the habitat and populations and alter the sheet flow dynamics of rain water runoff. Tortoise and other animal road kills are also an important source of food for ravens, probably facilitating raven survival and population increases, and as such, roads are another indirect source of mortality to tortoises.

Utility Corridors. Corridors formed by utility and energy rights-of-way cause linear impacts to tortoise populations and may have far reaching impacts well beyond those of many point sources of impacts (e.g., developments). Far more tortoise home ranges are traversed by a narrow linear corridor than by a more condensed non-linear project of similar acreage. Further, Olson (1996) reported that the construction of a natural gas pipeline had the greatest impact on tortoises and habitat, construction of a transmission line had intermediate impacts, and a fiber optic line was the most benign. Of 53 tortoises reported accidentally killed during implementation of 171 biological opinions in California and Nevada between 1989 and 1995, 41 of them (77%) were found dead on two linear projects, including the Mojave-Kern Pipeline and Meade-Adelanto Transmission Line (Circle Mountain Biological Consultants. 1996).

Considerable habitat destruction or alteration occurs when pipelines and power lines are constructed, and the impacts are repeated as maintenance operations, new pipelines, or new power lines are placed along existing corridors. Pits left open for pipe installation or maintenance may serve as traps for tortoises and other animals (Olson et al. 1993). The habitat conversions during early stages of post-construction succession not only preclude use by tortoises, but may function to inhibit or reduce dispersal across the corridor, thus effectively fragmenting a previously intact population. Furthermore, the presence of utility towers in areas otherwise devoid of other raven nesting substrates (e.g., joshua trees, palo verdes, cliffs), may introduce heavy predation to an area previously largely immune to this activity.

Low-level Impacts. Several additional anthropogenic activities also impact tortoise populations, but may be of lesser concern because the amount of area (hence, number of tortoises) impacted is small, total effect or probability of effect on impacted animals is low, our knowledge of the potential effects is low, or our ability to control them is largely non-existent. These include: illegal collecting, energy and mineral development, uncontained refuse, handling and manipulating tortoises, noise, non-motorized recreation, and vandalism. Although many tortoises probably die from it (Peterson 1994, 1996) drought is also considered a low level of threat because, although it may confound the effects of anthropogenic factors, it is a natural phenomenon and there is virtually nothing that can be done directly to minimize its effects.

Biological Standards: Based on a series of Population Viability Analyses (PVA), the recovery plan for the desert tortoise (USFWS 1994) recommended that several areas of approximately 1000 mi^2 (1610 km^2) of tortoise habitat be conserved and managed for tortoise recovery. A PVA is a process that uses information on the special genetic and demographic traits of small populations to predict the probability of extinction over a given period of time. It is wise to set recovery goals (target number of individuals) that have relatively low probabilities of extinction. These analyses resulted in a series of conditions for the viability of populations for a period of 500 years (20 tortoise generations), and were based on several conditional assumptions.

Condition 1. A genetically viable population of desert tortoises must be composed of at least 5000 adults. This condition is based on two major assumptions: 1) 500 individuals (adults) of any species must actively and successfully reproduce and pass genes onto the next generation to maintain sufficient genetic heterogeneity (Franklin 1980, USFWS 1994; cf. Dawson et al. 1986, Lande and Barrowclough 1987) and 2) only 10% of the adult population actually contributes to future generations (Ryman et al. 1981, Shull and Tipton 1987, USFWS 1994).

Condition 2. A demographically viable population of desert tortoises must be composed of at least 50,000 adults to cushion it against environmental stochasticity (variation in population growth rates). This condition is based on several assumptions (USFWS 1994): 1) the average growth rate for tortoise populations is 0.985 (which actually represents an overall decline and is based on data collected between 1979 and 1989 from 13 tortoise populations throughout the Mojave Desert), 2) standard deviation in annual growth rate is 0.096 (20% higher than measured between 1979 and 1989 to account for greater observed and hypothesized variation than existed during those ten years), 3) environmental and population conditions (other than variance in population growth rates) between 1979 and 1989 are applicable for the next 500 years. It should be noted that catastrophes (e.g., disease, drought, major habitat destruction) were not included, the 20% increase in variation in population growth rate resulted in 250% increase in the minimum viable population size estimated, and two of the PVAs indicated that increasing population growth rates (by reducing mortality or increasing reproduction) to near 1.0 are very important for raising the probability of population persistence.

Condition 3. A viable population of desert tortoises must maintain an average minimum density of 10 adults per mi^2 (6 per km^2). This minimum number ensures that adults have ample opportunity to encounter likely mates. This condition assumes that 1) space requirements for finding a mate can be based on years of greatest home range sizes (years with low forage production) and is approximately 125 acres (50 hectares), 2) male home ranges show little overlap with neighboring males, and 3) adult male to adult female ratio is 1:1. If these assumptions are met, every female is likely to encounter a male at least in years of low forage production.

Condition 4. Each reserve should contain a minimum of 1000 mi^2 (1610km^2) of tortoise habitat. This condition assumes that 1) 5000 adult tortoises are required to maintain a genetically viable population (Condition 1, above), 2) adult tortoises must exist at an average density of 10/mi^2 (6/km^2; Condition 3, above), and 3) tortoise habitat and presence are patchy, thus some areas would contain lower densities of tortoises. This space requirement is inadequate to ensure viability based on demographic considerations (Condition 2, above), which requires 5000 - 10,000 mi^2 (8050-16,100 km^2) of tortoise habitat and as such is optimistic and makes recovery dependent on Condition 5, below.

Condition 5. Modern principles of preserve design should be employed when developing reserves. This condition assumes that populations are at a lower risk of

extinction when: 1) reserves have low perimeter relative to area (i.e., approaching circular shape), 2) there is low fragmentation of reserves (hence, populations), 3) dispersal occurs among multiple reserves, and 4) there are two or more reserves for the species. The Recovery Plan recommends that there be three reserves (Desert Wildlife Management Areas, DWMAs) in the West Mojave Desert, each containing approximately 1000 mi^2 (1610 km^2) of tortoise habitat and that areas of suitable habitat be maintained between the three DWMAs to facilitate dispersal, thereby reducing the probability of extinction of any one population. If 1000 mi^2 (1610 km^2) of contiguous tortoise habitat cannot be included in a single reserve, then smaller segments, the sum of each should total at least 1000 mi^2 (1610 km^2), should be connected by corridors of usable tortoise habitat. Reserves that adhere less to these principles should be more strictly managed to reduce mortality and increase reproduction.

Condition 6. Population growth rates of 1.0 (stable population) should be achieved and maintained by reducing levels of mortality. This is particularly important in reserves where total population sizes are less than 10,000 - 20,000 animals. This condition assumes that: 1) starting populations are sufficiently large (e.g., much larger than 2,000 and probably closer to 20,000 adults), 2) variation in population growth rates does not increase greatly, and 3) no catastrophes occur. The more mortality is reduced through strict management, the greater will be the population growth rate and will be the required space of reserves. The converse is also true, the less mortality is reduced, the larger the reserves have to be to maintain the same probability of persistence. However, if population growth rates are 0.975 or less, it becomes highly unlikely that the population will persist for more than 400 years.

The assumptions made above concerning number of tortoises or amount of space are estimates based on the best available data. Actual numbers and areas necessary to establish and maintain viability are unknown, but the PVAs provide the only realistic means available for determining quantifiable targets for management areas.

Mortality can be reduced and reproduction increased through maintenance and implementation of several measures designed to reduce the effects of the threats listed above. Following is a list of actions recommended in the Recovery Plan (USFWS 1994). Most actions are oriented towards reserve-level management within DWMAs, but some apply to activities outside of the DWMAs. Activities that should be prohibited within DWMAs include: vehicles driving off of designated routes, competitive and organized ORV activities, habitat-destructive military maneuvers, clearing for agriculture, clearing for new landfills, surface disturbances that diminish habitat, livestock grazing (but perhaps allow experimental grazing in some non-core

areas), feral burros and horses, vegetation harvest (w/o permit), collecting of biological specimens (w/o permit), dumping and littering, deposition of captive or displaced tortoise & other animals (w/o permit), uncontrolled dogs, discharge of firearms (except for hunting between Sept. and Feb.). Some of these activities could possibly be reintroduced once scientific research provides hard evidence that specific activities, when properly managed, have minimal impact on tortoise population viability.

The following additional actions should be implemented: erect barrier fences and passageways along selected roads and highways, sign and in some cases fence boundaries near communities and Open Areas, reduce raven predation on juvenile tortoises, implement translocations from adjacent areas, designate Ord-Rodman DWMA as an Ecological Reserve and Research Natural Area, establish drop-off site in Barstow for adopting captive tortoises, remove of feral dog packs, initiate semi-wild breeding program once it has been shown with scientific evidence that captive release can be successful and is necessary, establish visitor and resident education center and programs.

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MOJAVE FRINGE-TOED LIZARD
Uma scoparia

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Management Status: Federal: BLM Sensitive
California: Species of Special Concern (CDFG, 1998)

General Distribution:

The Mojave Fringe-toed Lizard (*Uma scoparia*) is endemic to southern California and a small area of western Arizona, where it is restricted to aeolian sand habitats in the deserts of Los Angeles, Riverside, and San Bernardino Counties in California and La Paz County in Arizona (Van Denburgh, 1922; Smith, 1946; Schmidt, 1953; Norris, 1958; Pough, 1974; Stebbins, 1985). Nearly all localities are associated with present-day and historical drainages and associated sand dune complexes of the Mojave and Amargosa Rivers (Norris, 1958). Along the Amargosa River, this species is found at Ibex Dunes, north of Saratoga Springs, and at Dumont Dunes on the west slope of the Kingston Mountains, San Bernardino County (Norris, 1958). Along the present-day Mojave River it is found at the following localities: Peck's and Wilsona Butte, Los Angeles County; El Mirage Dry Lake, Harper's Dry Lake, Lenwood, Daggett, Yermo, Newberry Springs, Pisgah, Ludlow, the west slope of Alvord Mountain, Cronese Lake, Silver Lake, Crucero, Sands Siding, Devil's Playground, Coyote Lake, and Kelso Dunes, San Bernardino County (Norris, 1958; Mayhew, 1964b; de Queiroz, 1992). Along the Pleistocene discharge channel of the Mojave River, it is found at the following localities: Bristol Dry Lake, Cadiz Dry Lake, and Dale Dry Lake, San Bernardino County; Rice Valley, Pinto Basin, Palen Dry Lake, and Ford Dry Lake, Riverside County; and Bouse Dunes, 15 mi. (24.3 km) southeast of Parker, La Paz County (Norris, 1958; Pough, 1974).

Distribution in the West Mojave Planning Area:

This species is predominately distributed throughout the eastern portions of the WMPA. Localities within the WMPA include from west to east: Peck's and Wilsona Butte, Los Angeles County; El Mirage Dry Lake, Harper's Dry Lake, Lenwood, Daggett, Yermo, Newberry Springs, the west slope of Alvord Mountain, Pisgah, Cronese Lake, Crucero, Ludlow, and Dale Dry Lake, San Bernardino County (Norris, 1958; Mayhew, 1964b; de Queiroz, 1992).

Natural History:

The Mojave Fringe-toed Lizard (MFTL) is a medium-sized lizard (SVL 4.5 in [112 mm]) with a dorsoventrally compressed body and tail, small dorsal head and body scales, pointed snout in lateral profile, countersunk lower jaw, obliquely keeled supralabial scales,

large eyelid fringe scales, large anterior auricular scales, large imbricate shoulder and upper arm scales, greatly enlarged lamellar fringes on third and fourth hind-toe, a tail equal to body length, and two large postanal scales in males, which are only slightly enlarged in females. The dorsal ground color is light brown to yellowish, with dark ocelli pattern on the body, limbs and tail. The ventral color is light yellow to white, with one to three dark crescents across the throat region, a dark ventrolateral body blotch between the fore- and hindlimbs, and dark caudal bars on the posterior portion of tail (Van Denburgh, 1922; Heifetz, 1941; Stebbins, 1944, 1985; Smith, 1946; Norris, 1958; Pickwell, 1972; de Queiroz, 1989).

The MFTL is distinguished from all other species of fringe-toed lizards by the presence of crescent-shaped markings on the throat, a nasal process of the premaxilla bone with the lateral crests reduced posteriorly, and a frontonasal fontanelle commonly present in the skull (Cope, 1895; Heifetz, 1941; Schmidt and Bogert, 1947; Norris, 1958; de Queiroz, 1989). It can be further distinguished from the Coachella Valley fringe-toed lizard (*Uma inornata*) by the presence of a dark ventrolateral body blotch between the fore- and hindlimbs, dorsal ocelli that never reticulate to form a lineate pattern, and usually five internasals instead of three (Heifetz, 1941; Smith, 1946; Norris, 1958).

The MFTL has numerous adaptations associated with its highly arenicolous (= sand-dwelling) life style (Cope 1894; Van Denburgh, 1922; Mosauer, 1932, 1935; Stebbins, 1944, 1972; Norris, 1958, 1967; Pough, 1970; Carothers, 1986; Luke, 1986). The most notable, of which, are the enlarged, triangular shaped lamellar fringes on the third and fourth digit of the hindfoot that enable these lizards to achieve considerable speeds on the sand surface (Stebbins, 1944; Norris, 1958; Carothers, 1986). Other adaptations associated with burying in the sand include a countersunk lower jaw, valved nostrils, keeled supralabials, enlarged and imbricate shoulder scales, and a dorsoventrally compressed body (Stebbins, 1944; Smith, 1946; Norris, 1958; Carothers, 1986). In addition, the dorsal network of dark ocelli on a yellowish ground color make these lizards extremely cryptic on the sandy substrate, while their more distinguishing characteristics are concealed ventrally on their throat, sides, and tail (Stebbins, 1944; Smith, 1946; Norris, 1958).

The MFTL is omnivorous, feeding on dried seeds, flowers, grasses, leaves, insects, and scorpions (Van Denburgh, 1922; Miller and Stebbins, 1964; Minnich and Shoemaker, 1970, 1972). It is likely that the food preference shifts seasonally as in the Coachella Valley fringe-toed lizard (*Uma inornata*) where more plant material is consumed in spring when it is available and arthropods later in the year (Durtsche 1992, 1995; see also, Minnich and Shoemaker, 1970). Juveniles eat more arthropods than plants (Minnich and Shoemaker, 1970). In captivity, species of *Uma* have been known to be aggressive towards other lizards and occasionally eat them (Shaw, 1950)

Sexual maturity is reached between 2.5-2.75 in (65-70 mm) SVL, two summers after hatching (Jennings and Hayes, 1994). Adults exhibit a breeding coloration of a yellowish-green ventral wash that becomes pink along the sides between April and July (Mayhew, 1964a, 1964b; Stebbins, 1985). Courtship gestures include head bobbing and rapid, alternate, up and down waving of the front legs and feet (Carpenter, 1963; Mayhew, 1964a). Breeding activity occurs between April and July (Mayhew, 1964b). Females lay 1-5 eggs in hummocks or sandy hills during the months of May through July

(Stebbins, 1954, 1985; Kauffman, 1982). Hatchlings appear in September (Miller and Stebbins, 1964). More young are produced after wet winters (Mayhew, 1964a; Fromer et. al., 1983; Barrows et. al., 1995).

Males actively defend their home range which average 0.25 acres (0.10 ha; Kauffman, 1982). Aggression is displayed through a series of elaborate postures including lateral orientation and compression of the body, extension of the dewlap, and push-up displays (Carpenter, 1963, 1967). Juveniles do not defend territories until they become subadults (Jennings and Hayes, 1994). Seasonal activity occurs between March and October, with hibernation occurring between November and February (Mayhew, 1964a, 1964b). Daily activity patterns are temperature dependent (Miller and Stebbins, 1964). The MFTL has a internal body thermal voluntary maximum of 112.1 F (44.2 C), thermal voluntary minimum of 78.4 F (25.8 C), and thermal preference of 99.5 F (37.5 C; Mayhew, 1964b).

Predators of the MFTL include badgers (*Taxidea taxus*), coyotes (*Canis latrans*), hawks, shrikes, roadrunners (*Geococcyx californianus*), burrowing owls, leopard lizards (*Gambelia wislizenii*), and various snakes (Norris, 1958; Miller and Stebbins, 1964; Gracie and Murphy, 1986).

Habitat Requirements:

Mojave Fringe-toed Lizards are restricted to areas with fine, aeolian sand including both large and small dunes, margins of dry lakebeds and washes, and isolated pockets against hillsides (Stebbins, 1944, 1985; Smith, 1946; Norris, 1958). These areas are generally within creosote scrub desert between elevations of 300-3,000 ft (90-910 m; Norris, 1958; Stebbins, 1985). Sand dune ecosystems, including their source sand and sand corridors, are necessary for the long-term survivorship of aeolian sand specialists, such as, fringe-toed lizards (Barrows, 1996). Specific habitat requirements for the Coachella Valley fringe-toed lizard include access to shaded sand for thermoregulatory burrowing (Muth, 1991) and is likely required for the MFTL as well.

Population Status:

No data on population status and relative density of the MFTL is available. Tanya Trepanier, from the Royal Ontario Museum, Toronto, Canada, is currently conducting general population surveys and is studying the interpopulational relationships through the use of DNA sequence data. At the time of this report, her results have not been completed. Through personal communications, she reports that the MFTL is no longer found at a number of historical localities, while other populations appear to contain only a small numbers of individuals.

Threats Analysis:

The loose wind-blown sand habitat, upon which the MFTL is dependent, is a fragile ecosystem requiring the protection against both direct and indirect disturbances (Weaver, 1981; Beatley, 1994; Barrows, 1996). Potential direct disturbances include habitat loss or damage from urban development, off-highway vehicles (OHV), and agriculture. Potential indirect disturbances are associated with the disruption of the dune ecosystem source sand, wind transport, and sand transport corridors.

The decline of the closely related Coachella Valley fringe-toed lizard (*Uma inornata*) is attributed primarily to the urban development associated with the population increases of that region (Beatley, 1994; Barrows, 1996). Viable, long-term habitat has been reduced to 2% of the original range of the Coachella Valley fringe-toed lizard (Barrows et. al., 1995). For the Coachella Valley fringe-toed lizard , habitat loss is the result of land conversion to agriculture, roads, houses, and golf courses (Weaver, 1981; Beatley, 1994). Off-highway vehicles also have been implicated in habitat degradation (Beatley, 1994). Long-term habitat sustainability is further threatened by the disruption of dune building mechanisms. One such problem is the interruption of sand movement by buildings, railroad windbreaks, roads, and other man-made alterations (Weaver, 1981; Beatley, 1994).

Although there is no published data suggesting a decline in population sizes of the MFTL, similar urban development threats exist in the WMPA to cause concern that populations will be (or have already been) adversely affected.

Biological Standards:

Management efforts should be directed at identifying aeolian sand habitats and assessing the present condition of all known populations. Protected land should contain viable, long-term habitat, encompassing ecosystem-level processes that lead to the formation of these habitats. The physical mechanisms attributed to the formation of sand dunes should be integrated into management plans. Protected land should include areas for source sand, wind and sand corridors, as well as the sand dune habitat and its associated shade plants. Successful Habitat Conservation Plans (HCP) have been developed for the Coachella Valley Fringe-toed Lizard (The Nature Conservancy, 1986; Beatley, 1994; Barrows, 1996). These HCPs incorporated an ecological model to identify habitat with long-term viability, as well as those habitats deemed nonviable because their sand source and/or wind corridor were blocked by previous developments (Beatley, 1994; Barrows, 1996). Further details within the Coachella Valley Fringe-toed Lizard HCP should be consulted. Direct habitat degradation can be further avoided through land use restrictions. Restrictions should prevent the conversion of viable habitat to agriculture and the many forms of urban development and prohibit the use of OHVs.

Because little is known about the population biology of the MFTL, mark and recapture studies should be implemented in as many populations as possible to gather much needed data (see Muth, 1987, 1991). In addition, genetic studies focusing on interpopulational relationships could greatly increase our understanding about population associations and their relative distinctiveness from each other. These studies may prove particularly useful if mitigation of existing populations is needed. Until sand dune ecosystem and population data become available, a management plan and recommendations for long-term population viability are inconclusive.

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PANAMINT ALLIGATOR LIZARD
Elgaria panamintina

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Management Status: Federal: USFWS Species of Concern; BLM Sensitive
California: Species of Special Concern (CDFG, 1998)

General Distribution:

The Panamint Alligator Lizard (*Elgaria panamintina*) is endemic to California, where it is known only from 16 disjunct localities in the Panamint Mountains (Brewery and Limekiln Springs, Surprise Canyon, Pleasant Canyon), Nelson Mountains (Grapevine Canyon), Inyo Mountains (Daisy Canyon, Lime Hill), and White Mountains (Batchelder Spring, Marble Canyon, Tollhouse Spring, Westgard Pass) of Inyo County, California. Sight records (12) have been reported for the eastern Argus, Cosos, Panamint, Inyo, and White Mountains of Inyo and southeastern Mono counties, California (Stebbins, 1985; Macey, 1986; Papenfuss, 1986; Macey and Papenfuss, 1991; Jennings and Hayes, 1994; La Berteaux and Garlinger, 1998). The species may also occur in Benton and Queen valleys and in mountains of western Nevada (Banta, 1965; Macey and Papenfuss, 1991). The elevational range of the species extends from 2500-7500 ft (760-2290 m).

Distribution in the West Mojave Planning Area:

In the WMPA this species is known only from sight records from several valleys in the eastern Argus Mountains.

Natural History:

The Panamint Alligator Lizard is a large lizard (SVL 150 mm [6 in]) with a tail nearly twice its body length. It is distinguished from all other congeners by having 10 (rarely 12) keeled longitudinal dorsal scale rows, 44-47 (x=45.7) transverse dorsal scale rows, and the presence of 7 or 8 complete, well defined, brown to dark-brown crossbands on the body, exclusive of the tail. The ventral surface is white with gray spots at the center or edges of the scales forming irregular scattered blotches. There are 6 rows of weakly keeled scales on the tail (counted on the first whorl from the base of the tail that contains 20 scales). The iris of the eye is pale yellow. Young possess black crossbands on the body and tail and interspaces of pale orange-yellow, fading to white laterally. Both sexes are similar in color and pattern; as adults, males have a broader, more triangular head then female's (Stebbins, 1958, 1985).

Comprehensive studies and field data on the natural history of the Panamint Alligator Lizard are lacking. Various aspects of the natural history are based on incidental field observations (see Jennings and Hayes, 1994 and Banta et al., 1996) and infrequent monitoring of pitfall traps (Banta, 1962, 1963). Museum and literature records based on 24 specimens indicates that the species is active from April-October. Peak activity periods are in June with a decrease in activity and aestivation occurring during the months of July and August (Banta, 1963). Although typically active during the day, Panamint Alligator Lizards are nocturnal during the hot summer

months (Dixon, 1975). There seems to be a preference for relatively low air temperatures (15-24 C; Stebbins, 1958). Long (1972) recorded body temperature (23 C) and air temperature (23.5 C) from one individual observed in the field.

The Panamint Alligator Lizard spends much of its time foraging in thick brush and along talus slopes. It is rarely encountered in the field, but is occasionally observed basking on rocks in open areas, near thick vegetation (e.g., wild grape; Stebbins, 1973; Long in litt., 1972). The diet of the Panamint Alligator Lizard is unknown, although mealworms have been eaten in captivity (Stebbins, 1958; Banta and Leviton, 1961; Banta, 1963).

An adult female captured on 1 May 1959 contained 12 eggs (2.4-4.4 mm in diameter; Banta, 1963). Mating of two individuals in captivity was observed from 15-17 May (Banta and Leviton, 1961). In addition, data relevant to the Southern Alligator Lizard (*E. multicarinata*) suggests that Panamint Alligator Lizards may produce a second clutch in late summer (see Burrage, 1965).

Although there are no records of predation on the Panamint Alligator Lizard, it may be a prey item of the following vertebrate species: Coachwhip (*Masticophis flagellum*), Striped Whipsnake (*M. taeniatus*), Western Patch-Nosed Snake (*Salvadora hexalepis*), California Kingsnake (*Lampropeltus getula*), Long-Nosed Snake (*Rhinocheilus lecontei*), Red-Tailed Hawk (*Buteo jamaicensis*), Greater Roadrunner (*Geococcyx californianus*), Loggerhead Shrike (*Lanius ludovicianus*), and Coyote (*Canis latrans*; Marlow, 1988; Jennings and Hayes, 1994).

Habitat Requirements:

The Panamint Alligator Lizard probably had a greater distribution in the mesic Pluvial period of the Pleistocene (Good, 1988). Following the Pluvial retreat, accompanied by increased aridity and the formation of desert environments, it survived as a relict to occupy mesic canyons of the Inyo, Panamint, and White mountain ranges (Banta, 1963). The species occurs most frequently in canyons supporting riparian habitat and nearby permanent springs. It is less abundant in xeric habitats associated with rocky alluvium and boulder talus slopes. This riparian habitat is dominated by Red Willow (*Salix laevigata*), Arroyo Willow (*S. lasiolepis*), Virgin's Bower (*Clematis ligusticifolia*), Wild Grape (*Vitis girdiana*), Scarlet Monkey Flower (*Mimulus cardinalis*), and Southern Maidenhair Fern (*Adiantum capillus-veneris*). Decaying willows, branches, and layers of leaves (over a foot thick) cover the riparian floor. Bordering the riparian are rocky talus slopes and boulder strewn hillsides dominated by xeric adapted plant species such as Creosote Bush (*Larrea tridentata*), Wormwood (*Artemisia ludoviciana*), Shad Scale (*Atriplex canescens*), California Buckwheat (*Eriogonum fasciculatum*), Acton's Encelia (*Encelia virginensis*), Beavertail Cactus (*Opuntia basilaris*), and Barrel Cactus (*Ferocactus acanthodes*) (see Stebbins, 1958 and Banta, 1963). Specimens collected in the dry washes and along talus slopes of Grapevine and Daisy canyons, indicate that the species may occur over a wider geographic range within the Owens, Panamint, and Saline valley hydrographic basins (Banta et al., 1996).

Population Status:

No data on population status and relative density of the Panamint Alligator Lizard are available.

Threats Analysis:

A potential decline in Panamint Alligator Lizard populations may be attributed to the direct loss of riparian habitat. Although there are no baseline data that suggests a current decline in population numbers, habitat loss or alteration due to expanded mining operations, off-highway vehicle (OHV) activity, grazing (domestic and feral), and introduction of non-native invasive plant species (e.g., Tamarisk) could have serious adverse effects in riparian areas where this species occurs. Over collecting, which could also contribute to the species decline, is a secondary consideration. Direct threats to populations are the result of human activities and will likely increase in the future.

Of concern, are proposed mining operations and their impact on the mortality of Panamint Alligator Lizards. Construction or improvement of access roads in concert with increased vehicular traffic through or adjacent to "islands" of riparian habitat could threaten lizard populations. Proposed mining operations that alter or modify springs, seeps, and stream flow would be a direct threat to the hydrology within the species habitat.

The hydrophilic plant, Salt Cedar (*Tamarix ramosissima*), is a non-native invasive species that has naturalized along the western slopes of the White-Inyo Range (up to 6000 ft [1829 m]). Tamarisk has also become widespread in mesic desert habitat and is a serious threat to springs and seeps in the region (DeDecker, 1991).

Off-highway vehicle activity has increased significantly over the last 15 years and many OHV enthusiasts have resorted to wenching up steep canyon walls and washes to explore the Panamint Mountains. On a visit to the species' habitat in Surprise and Pleasant Canyons on 13-14 April 1996, extensive dirt bike activity and several high use trails (i.e., wenching) were observed (pers. obs.).

Illegal collecting using pitfall traps and coverboards may also threaten populations.

All localities where the Panamint Alligator Lizard is known to occur are restricted use areas with limited access and, with the exception of State Highway 168 in the vicinity of Cedar Flats and Westgard Pass (Dixon, 1975), threats related to highway mortality and urban development are negligible.

Biological Standards:

It is critical that management efforts be directed at protecting habitats associated with all known Panamint Alligator Lizard localities. Specifically, impacts on hydrology should be avoided to ensure the long-term viability of populations. Off-highway vehicle activity and proposed mining operations are serious threats to core population centers. Protection of this species for future field studies can be achieved through land use restriction efforts. A well supervised, repetitive pitfall monitoring program could provide baseline data that would aid in our understanding of the distribution, habitat requirements, and populations status of the Panamint Alligator Lizard. Until additional distribution and population data become available, a management plan or recommendations for long-term population viability is inconclusive.

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SAN DIEGO HORNED LIZARD
Phrynosoma coronatum blainvillii

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Management Status: Federal: USFWS Species of Concern
California: Species of Special Concern (CDFG, 1998)

General Distribution:

The San Diego Horned Lizard (*Phrynosoma coronatum blainvillii*) is endemic to southern California and northern Baja California, México. In California, this species is distributed predominately throughout cismontane regions of the Transverse Ranges in Kern, Los Angeles, Santa Barbara, San Bernardino, and Ventura Counties, southward to the Peninsular Ranges in Orange, Riverside, and San Diego Counties (Van Denburgh, 1922; Smith, 1946; Reeve, 1952; Schmidt, 1953; Pickwell, 1972; Jennings, 1988; Jennings and Hayes, 1994; Brattstrom, 1997).

Distribution in the West Mojave Planning Area:

This species is distributed along the southern margin of the WMPA. Within the WMPA, the San Diego Horned Lizard (SDHL) occurs from the Antelope Valley California Poppy State Reserve eastward along the base of the San Gabriel and San Bernardino Mountains to Joshua Tree National Park (Jennings and Hayes, 1994; Brattstrom, 1997). The northernmost distribution records within the WMPA occur in the Antelope Valley California Poppy State Reserve (Brattstrom, pers. comm.). The population along the Mojave River, near Oro Grande, is believed to be extinct (Jennings and Hayes, 1994).

Natural History:

The SDHL is a medium-sized lizard (SVL 4.3 in [110 mm]) with a dorsoventrally compressed body and short tail. The posterior margin of head has enlarged head scales, orientated posteriorly, forming a crown of dagger-like spines; the two posteriormost spines are larger than those more lateral. The supraocular scale is pointed and orientated posteriorly, the dorsal body and legs are interspersed with enlarged, keeled to pointed scales, and the lateral body and tail have enlarged, pointed fringes. The dorsal ground color is yellowish, light brown, gray, white, or reddish-brown with dark blotches on the neck and back, usually matching the substrate color, while the ventral color is light yellow to white with dusky spots (Van Denburgh, 1922; Smith, 1946; Reeve, 1952; Stebbins, 1972, 1985).

The SDHL is distinguished from all other species of horned lizard by the presence of two rows of lateral body fringe scales, two to three rows of enlarged, pointed scales on

the sides of the throat, and smooth, convex dorsal head scales (Smith, 1946; Reeve, 1952; Stebbins, 1985; Brattstrom, 1997). Recent taxonomic studies of *Phrynosoma coronatum* (Grismer and Mellink, 1994; Brattstrom, 1997) indicate that there is no character basis for the recognition of the various subspecies.

The SDHL is usually a solitary animal that relies on camouflage in open areas and is known to bury itself in fine, loose soil (Stebbins, 1985; Jennings and Hayes, 1994). Defensive behavior includes lowering the head to expose the head spines (Smith, 1946; Reeve, 1952), puffing up the body (Tollestrup, 1981; Jennings and Hayes, 1994), and squirting of blood from the eye due to the rupturing of a sinus vessel inside the eyelid (Bryant, 1911; Klauber, 1939; Burleson, 1942; Smith, 1946; Reeve, 1952; Stebbins, 1985).

The SDHL is insectivorous, feeding primarily on native harvester ants (*Pogonmyrmex* sp.), but it will also feed on termites, beetles, flies, wasps, and grasshoppers (Ingles, 1929; Reeve, 1952; Miller and Stebbins, 1964; Dixon, 1967; Pianka and Parker, 1975; Stebbins, 1985; Jennings and Hayes, 1994).

Sexual maturity is reached at a size of 3 in (73-76 mm) SVL, two to three years after hatching (Howard, 1974; Pianka and Parker, 1975; Goldberg, 1983; Stebbins, 1985; Jennings and Hayes, 1994). A clutch of 6-17 eggs are laid between May and early July (Howard, 1974; Goldberg, 1983; Stebbins, 1985; Jennings and Hayes, 1994). Eggs hatch in approximately two months appearing in July and early August (Shaw, 1952; Howard, 1974; Goldberg, 1983; Jennings and Hayes, 1994).

Seasonal activity occurs between late March and early October, with hibernation setting in as early as August (Pequegnat, 1951; Howard, 1974; Jennings, 1987; Hager, 1992). Daily activity patterns are temperature dependent and lizards will emerge from their burial sites before sunrise to position themselves for basking in the first rays of sun (Heath, 1965; Hager, 1992; Jennings and Hayes, 1994). The SDHL has a internal body thermal voluntary maximum of 102.2 F (39.0 C), thermal voluntary minimum of 69.4 F (20.8 C), and thermal preference of 94.8 F (34.9 C; Brattstrom, 1965; Heath, 1965; Jennings and Hayes, 1994).

Predators of the SDHL include coyotes (*Canis latrans*), badgers (*Taxidea taxus*), foxes, kestrels, falcons, shrikes, roadrunners (*Geococcyx californianus*), burrowing owls, and various snakes including the southern pacific rattlesnake (*Crotalus viridis helleri*) and Striped Racer (*Masticophis lateralis*; Bryant, 1916; Von Bloeker, 1942; Klauber, 1972; Eakle, 1984).

Habitat Requirements:

San Diego Horned Lizards are found in a wide variety of habitats including coastal sage scrub, chaparral, grassland, coniferous forest, oak woodland, riparian, and the margins of the higher elevation desert where it is restricted to the juniper-desert chaparral (Grinnell and Grinnell, 1907; Van Denburgh, 1922; Klauber, 1939; Smith, 1946; Dixon, 1967; Stebbins, 1985; Jennings and Hayes, 1994; Brattstrom, 1997). Within each of these habitats, this species prefers areas with loose, fine soils, an abundance of open areas for basking, and plenty of native ants and other insects (Jennings and Hayes, 1994). This species has been reported from elevations ranging from sea level to 8,000 ft (0-2600 m; Brattstrom, 1997).

Population Status:

No reliable data on population status and relative density of the SDHL are available. Hager (1992) presented information on home range and movement in San Bernardino and Riverside Counties, but due to difficulties in re-sightings, home ranges are likely underestimated and interpretations of movement patterns inconclusive (Jennings and Hayes, 1994). The SDHL is believed to be extinct in 45% of its original range in southern California, including desert regions near Palmdale, Los Angeles County and the Mojave River, San Bernardino County (Jennings and Hayes, 1994).

Threats Analysis:

A number of factors have been implicated in the decline of the SDHL, including collecting, habitat loss, off-highway vehicles (OHV), livestock grazing, and the introduction of Argentine ants (Jennings and Hayes, 1994). Unfortunately, there is little baseline data to properly understand the exact nature of the current decline.

The SDHL was heavily exploited at the turn of the century for the curio trade (Jennings, 1987); horned lizards were varnished or sold as pets (Klauber, 1939). Later, biological supply companies and the modern pet trade contributed to their exploitation, until 1981, when commercial collecting was banned (Jennings and Hayes, 1994).

The SDHL's habitat is increasingly becoming destroyed and fragmented due to urban development, the conversion of land to agriculture, and OHV use (Jennings and Hayes, 1994). Habitat loss and collecting have been cited as the main reasons for this species decline (Jennings, 1987). Additional disturbances may include the construction of fire breaks and the use of prescribed burning.

Other pressures include the loss of the native ant food base due to their progressive elimination by Argentine ants (*Iridomyrmex humilis*; Ward, 1987; Jennings and Hayes, 1994) and increased predation by domestic dogs and cats (Hayes and Guyer, 1981; Jennings and Hayes, 1994).

Biological Standards:

Management efforts should be directed at identifying the best remaining habitat and largest populations to determine areas that should be protected from human disturbance (Jennings and Hayes, 1994). Until then, some level of protection can be achieved through land use restrictions. Extensive studies are needed to develop baseline data about existing populations. The impact of OHV activity, predation by domestic pets, and the replacement of native ants by introduced Argentine ants need to be more precisely understood, as well as the effects of livestock grazing and prescribed burning (Jennings and Hayes, 1994). A well-supervised, repetitive pitfall trap program could provide baseline data that would aid in our understanding of the distribution and population status of the SDHL. In addition, genetic studies focusing on interpopulational relationships could greatly increase our understanding about population associations and their relative distinctiveness from each other. These studies may prove particularly useful if mitigation of existing populations is needed. A management plan or recommendation for long-term population viability is inconclusive until additional population data become available.

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WESTERN POND TURTLE
Clemmys marmorata

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Management Status: Federal: USFWS Species of Concern; BLM Sensitive
California: Species of Special Concern (CDFG, 1998)

General Distribution:

The western pond turtle (*Clemmys marmorata*) formerly ranged from extreme western Washington and British Columbia to northern Baja California, mostly to the west of the Cascade-Sierra crest (Ernst et al., 1994). Disjunct populations exist, or existed, in the Truckee, Humboldt, and Carson Rivers in Nevada, Puget Sound, and the Columbia Gorge. Other isolated, but extant, populations are found in the interior draining Mojave River of California at least as far into the Mojave Desert as Afton Canyon, and in the Amargosa River, Los Angeles County. Fossils from the Camp Cady area establish the presence of western pond turtles in the Mojave Desert to at least the Pliestocene (Jefferson, 1968). Scattered records, including the individual reported by Jennings and Hayes (1994) in Andreas Canyon near Palm Springs, California, likely represent introduced specimens (Holland, 1994).

Distribution in the West Mojave Planning Area:

Records are scattered along much of the Mojave River including Yermo and Victorville (Seeliger, 1945). Brattstrom and Messer (1988) speculated that some turtles remain in Deep Creek and reported previous records from the Mojave Narrows near Victorville, and Afton Canyon. The author observed a western pond turtle in a beaver (*Castor canadensis*) pond along the Mojave River below the Victorville sewage treatment plant in 1998. A population exists at Camp Cady and several specimens from there were transplanted to ponds at the Desert Studies Center at Zzyxx (outside the planning area), and the California Desert Information Center (CDIC) in Barstow. It is assumed that the Zzyxx transplant failed, but a few turtles remain at CDIC and have produced offspring. Some of the turtles at Camp Cady may have been introduced from Carbon Canyon, Orange County in the 1970's, but this claim has not been positively substantiated. The author initiated studies on the ecology of western pond turtles in the Mojave River in 1998 and over 30 specimens are now marked at Camp Cady and Afton Canyon. Several recent records (1990, 1995) are listed in the Natural Diversity DataBase for the southwestern Antelope Valley near Elizabeth Lake and the Amargosa River, Los Angeles County. No published literature is available to ascertain the status of these populations but studies have been initiated and a population of over 50 turtles are now marked (David Muth, pers. comm.).

Natural History:

The western pond turtle is a small turtle with a relatively low carapace. The shell may exhibit a pattern of dark spots or lines that radiate from the centers of the scutes, or it may be almost patternless olive brown, dark brown, or grayish. The oval carapace is up to 8.3 inches (21 cm) in large individuals (Holland, 1994). On the Mojave River, at Camp Cady and Afton Canyon,

males have an average carapace length of 5.2 inches (13.1 cm), while females are slightly larger at 5.7 inches (14.5 cm) (Lovich, unpublished data). In the Amargosa River (Los Angeles County), average carapace length of males and females is 5.0 inches (12.8 cm) and 5.4 inches (13.7 cm), respectively (David Muth, unpublished data).

The pale yellow plastron is hingeless and may have dark blotches along the rear margins of the scutes. The skin is gray, with some pale yellow on the neck, chin, forelimbs, and tail. The head is plain or reticulated. Males have a concave plastron, lighter throat coloration, and the anal vent situated posterior to the rim of the carapace.

Two subspecies are recognized. The northwestern pond turtle (*Clemmys marmorata marmorata*) ranges south to San Francisco Bay, and east to Nevada. A pair of well-developed triangular inguinal scutes on the bridge characterizes it, and its brown or grayish neck and head are well marked with dark dashes. The throat is pale in contrast with the sides of the head. The southwestern pond turtle (*Clemmys marmorata pallida*) is found south of San Francisco Bay including the West Mojave Planning Area (WMPA). It is identified by its poorly developed inguinal scutes (absent in 60% of individuals), and by the uniform light color of the throat and neck. The two races intergrade over a large area in central California (Bury, 1970; Stebbins, 1985).

Although the validity of recognized subspecies has been questioned, recent genetic analysis using DNA fingerprinting supports their distinctiveness (Gray, 1995; Janzen et al., 1997). Holland (1992) suggested that there are actually three species in what is currently recognized as *Clemmys marmorata* including a northern species, a southern species (which would include those in the WMPA), and an undescribed species from the Columbia River. Additionally, Janzen et al. (1997) observed several unique genetic variants in southern California and Baja California, suggesting that special care should be taken to preserve and manage these populations. In fact, southern populations may be distinct enough to warrant recognition as separate species (Janzen et al., 1997). However, no formal taxonomic revisions have been published and all populations continue to be recognized as *C. marmorata*. Holland (1992) further suggested, based on preliminary analysis, that turtles in the Mojave River showed a high level of morphological differentiation from other populations in southern California.

Nothing has been published on the ecology of western pond turtle populations in the WMPA, but considerable information has accumulated for the species elsewhere in its range (Ernst et al., 1994; Holland, 1994). The following narrative borrows heavily from these sources and incorporates the results of other studies. Seasonal activity varies geographically and turtles may be active in every month at some localities (Holland, 1994). In the northern portion of its range, western pond turtles are active primarily from February-November (Evenden, 1948; Bury, 1972). In northern California it starts to forage early in the morning and then basks intermittently thereafter with most basking occurring from 0900-1000. During the summer, turtles may forage in the late afternoon or early evening. Western pond turtles in northern California apparently avoid body temperatures over 95° F (34° C), usually terminating aerial basking at about 90-95° F (32-34° C), well below its critical thermal maximum of 104° F (40° C). Most normal activities occur at body temperatures from 75-90° F (24-32° C; Bury, 1972).

Adult males have larger home range sizes and lengths than do adult females and these in turn are larger than those of juveniles. In a population congregated in pools in a northern California stream, male home range size averaged 2.42 acres (0.98 ha) with a mean length of 3,201 feet (976 m). For adult females the figures were 0.62 acres (0.25 ha) and 813 feet (248 m).

Values for juveniles were 0.89 acres (0.36 ha) and 1,191 feet (363 m) (Bury, 1972). During the summer these turtles moved from pool to pool within the stream system; 53.5% of males, 27.5% of females, and 32.6% of juveniles moved upstream, while 34.0% of males, 38.9% of females, and 27.1% of juveniles traveled downstream. The average male moved 1,161 feet (354 m) during Bury's (1972) study, while the average female and juvenile moved only 554 feet (169 m) and 466 feet (142 m), respectively. More than 81% of the males moved over 656 feet (200 m), but only 36.6% of females and 22.5% of juveniles traveled a greater distance than 656 feet (200 m); 26.6% of the males, 6.1% of females, and 6.2% of juveniles migrated over 1,640 feet (500 m). In three years, the average male moved approximately 2,680 feet (817 m). Turtles studied by Bury were capable of moving distances of at least 1 mile (1.6 km) overland to adjacent water bodies and later returning, but Holland (1994) reported overland movements of up to 3.1 miles (5 km). At another site in coastal southern California, four radio-equipped females averaged daily movements of 92 feet (28.0 m), 179 feet (54.6 m), 198 feet (60.5 m), and 286 feet (87.1 m), respectively during the period 20 May to 21 June 1989 (Rathbun et al., 1992). Small movements between adjacent ponds and wetlands have been observed on several occasions at Camp Cady and Afton Canyon, both on the Mojave River (Lovich, unpublished).

Basking western pond turtles engage in a variety of aggressive behaviors to ensure adequate spacing at basking sites (Bury and Wolfheim, 1973). Some turtles bite or ram to push the other turtle off its perch. Most basking aggression takes place before noon (0900-1159 hours).

Bury (1972, 1989) estimated a population density of 529 turtles/acre (214 turtles/ha) and a biomass of 123 pounds/acre (137 kg/ha) in a 2.2 mile (3.5 km) stretch of stream in northern California. Turtles were not distributed uniformly and the greatest densities occurred in pools. Each meter of stream deeper than 1.6 feet (0.5 m) had an estimated 0.6 turtles. The adult sex ratio was 1.7 males for every female and thirty-five percent of the population was composed of juveniles. A sex ratio of about 1:1 was reported by Goodman (1997a) for a population in the West Fork of the San Gabriel River in southern California. The adult sex ratio at Camp Cady and Afton Canyon, both on the Mojave River, is biased at 1.3 males per female (Lovich, unpublished). Under optimal conditions in Oregon population density may approach 202 turtles/acre (500 turtles/ha; Holland, 1994).

The reproductive biology of western pond turtles is poorly known. Females begin laying eggs at a carapace length greater than 4.3 inches (11 cm; Goodman, 1997a,b) and are probably 6-7 years old (Holland, 1994). Courtship and mating have been observed in the field during most of the year except December-January (Holland, 1988; Buskirk, 1991; Goodman, 1997a). Nesting extends from late April through August (Holland, 1994), depending on the latitude, with a peak from late May to early July. At Camp Cady, on the Mojave River, females nest in late May and early June (Lovich, unpublished). Females may travel along a waterway as far as 1.2 miles (2 km) to distant nesting areas if suitable nesting habitat is not available locally (Rathbun et al., 1992). Nests are excavated in either the morning or evening (Storer, 1930) and are usually located along stream or pond margins. However, nests may be located over 328 feet (100 m) from the water on hillsides. All six terrestrial locations where Rathbun et al. (1992) found a radio-equipped female during the nesting season were in open, grassy areas with a southern exposure, as is typical for the species (Holland, 1994). Her sojourns onto land began between 1700 and 2000 hours, and she spent the night ashore, returning to the water the next morning between 0815 and 0900 hours. A second female also made overnight trips to a similar habitat, presumably to nest. Three flask or

pear-shaped nest cavities examined by Rathbun et al. (1992) were 2.6-3.1 inches (6.5-8.0 cm) deep with a 2.6-2.8 inch (6.5-7.0 cm) wide egg chamber and a 1.4-1.6 inch (3.5-4.0 cm) mouth. Nesting forays may require several days on land (Holland, 1994).

Clutch size ranged from 1-13 eggs (mean 6.12) for 168 clutches examined by Holland (1994) and is significantly correlated with body size (Holland, 1994; Goodman, 1997a). Clutch size for 5 gravid females at Camp Cady ranged from 4-6 eggs with an average of 4.8 (Lovich, unpublished). Mean egg width is also possibly correlated with body size (Goodman, 1997b). Western pond turtles are known to produce up to two clutches per year in the Los Angeles Basin of southern California. Goodman (1997b) observed 3 out of 7 females double clutching in one year. First clutches were oviposited between 4 and 14 May and second clutches were oviposited between 10 and 20 June. The internesting intervals between successive clutches were 38, 38, and 41 days. Not all females laid eggs in every year: 1992 - 3 of 8 were gravid, 1993 - 7 of 14 were gravid, and 1994 - 3 of 9 were gravid. Of 15 females of potential reproductive size (carapace length>4.3 inches or 11 cm) studied, only 2 clutched in two consecutive years and only one clutched in three consecutive years. First and second clutch sizes produced in the same year did not differ significantly within individual females, nor did mean egg width.

The natural incubation period is 80-126 days and varies with latitude (Goodman, 1997a; Holland, 1994). Eggs incubated at 77-91° F (25-33° C) by Lardie (1975) and Feldman (1982) hatched in 73-81 days. Feldman (1982) noted that hatchlings did not leave the egg if the temperature exceeded 81° F (27° C), but once moved to a cooler environment emerged within 2-3 hours. Western pond turtles have environmental sex determination with males produced at low incubation temperatures and females at high temperatures. The pivotal temperature is approximately 86° F (30° C; Ewert et al., 1994). Hatching success averages 70% but complete failure of nests is not uncommon in some years or locations (Holland, 1994). Goodman (1997a) reported an 80% hatching success rate for 15 eggs in 3 nests in southern California. In southern California, most hatchlings emerge in the early fall, while some overwinter in the nest. In northern California and Oregon hatchlings remain in the nest through the winter (Holland, 1994). Hatchlings are 0.9-1.2 inches (23-31 mm) in carapace length (Holland, 1994).

Growth rates for hatchlings average 0.13 inches/month (3.29 mm/month) during the first season of growth and then decline to 0.08 inches/month (1.95 mm/month), 0.03 inches/month (0.64 mm/month), and 0.04 inches/month (0.89 mm/month) during the second, third, and fourth growing seasons, respectively. After the fourth growing season growth rates slow to about 0.4 mm/month (Holland, 1994). Age can be estimated in western pond turtles up to about 16 years as deposition of scute rings on the shell is essentially annual (Bury and Germano, 1998). Annual survivorship of 1-3 year age classes is estimated to be 10-15%. Annual mortality of adults averages 3-5% (Holland, 1994).

Western pond turtles eat a wide variety of food items. Known foods include: algae, various plants (including the pods of the yellow water lily), snails, crustaceans (crayfish, *Daphnia*), isopods, insects, fish, frogs (tadpoles and adults), mallard duck carrion, and a mouse fragment (Pope, 1939; Evenden, 1948: Carr, 1952; Holland, 1985; Bury, 1986). Prey size and proportions of prey items differs in the diets of males, females and juveniles. Males consume more insects and vertebrates than do females, who eat more algae, and males seem to prefer larger food items. Juveniles eat smaller foods and take higher numbers of individual prey than do adults (Bury, 1986). Goodman (1998) observed a juvenile western pond turtle feeding on a coyote (*Canis latrans*) scat in a southern California stream.

Archaeological sites in California have produced remains of western pond turtles that may have been eaten by native Americans (Schneider and Everson, 1989).

Habitat Requirements:

The western pond turtle occupies a wide variety of wetland habitats including rivers and streams (both permanent and intermittent), lakes, ponds, reservoirs, permanent and ephemeral shallow wetlands, abandoned gravel pits, stock ponds, and sewage treatment lagoons (Holland, 1994). In streams, pools are the preferred habitat (Bury, 1972). Alteration of channel morphology and flow rates associated with dam construction diminished the quality of habitat for western pond turtles in some tributaries of the Trinity River in northern California (Reese and Welsh, 1998).

Western pond turtles have been collected from brackish estuarine waters at sea level to over 6,717 feet (2,048 m), but the species is uncommon above 5,015 feet (1,529 m; Stebbins, 1954; Bury, 1963; Holland, 1994). Optimal habitat seems to be characterized by the presence of adequate emergent basking sites, emergent vegetation, and the presence of suitable refugia in the form of undercut banks, submerged vegetation, mud, rocks and logs (Holland, 1994).

Holland and Goodman (1996) reported an aggregation of 19 western pond turtles in a crevice of granitic rock near a stream in San Luis Obispo County, California on September 26. The availability of suitable terrestrial shelter sites is necessary to provide protection from predators and thermal extremes. Overwintering and estivation sites are typically located in upland areas and in southern California may be over 197 feet (60 m) from water (Goodman, 1997a).

Population Status:

Populations are declining in southern California and over most of their northern range. Habitat destruction seems to be the major cause of its decline (Brattstrom, 1988; Brattstrom and Messer, 1988). Today only northern California and southern Oregon support extensive populations.

The decline of the western pond turtle in southern California has been recent and rapid. In 1960 there were 87 known localities for the species from Ventura County to the Mexican border. As of 1970, these were reduced to 57. In 1987, 255 sites were inspected of which 53 possessed turtles, and 25 of these were in Ventura County. Of the 53 sites, only 10 were thought to contain reproductively viable populations. South of the Santa Clara River, sites with western pond turtle populations become increasingly rare: Los Angeles County - 10, San Diego County - 8, Orange County - 4, western Riverside County - 3, and southwestern San Bernardino County - 3. Only five of the populations south of the Santa Clara River were thought to be reproductively viable (Brattstrom, 1988; Brattstrom and Messer, 1988).

Populations along the Mojave River were considered to be “small” by Brattstrom and Messer (1988). Holland (1991) estimated that no more than 100 western pond turtles are found in the Mojave River and noted that prospects for re-establishment in the event of extirpation are essentially zero. Ongoing research indicates that at least 34 western pond turtles survive at Camp Cady and Afton Canyon, combined (Lovich, unpublished).

In 1992 the Fish and Wildlife Service was petitioned to consider the species for listing under the provisions of the U.S. Endangered Species Act of 1973 (Fish and Wildlife Service, 1992). After formal review the Service declined to list the species. However, they reclassified

both recognized subspecies as category 2 candidates for listing (Fish and Wildlife Service, 1993), a designation that was later abandoned for all candidate species.

In a recent review of the status of California's amphibian and reptile species of special concern, Jennings and Hayes (1994) recommended that western pond turtles be reclassified as endangered from the Salinas River south along coast, and from the Mokelumne River south and inland in the San Joaquin river basin. Elsewhere in the state, they recommended threatened status.

Threats Analysis:

The greatest single threat to this species is habitat destruction. Over 90 percent of the wetland habitats within the historic range of the species in California alone have been eliminated due to agricultural development, flood control and water diversion projects, and urbanization (Fish and Wildlife, 1992). Associated with these threats has been an increase in habitat fragmentation and its attendant affects on genetic variability discussed below.

Lack of genetic variability may be a significant threat to the continued survival of populations in Oregon and Washington. The genetic similarity of these populations reflects a lack of dispersal and gene flow, and is probably a consequence of habitat fragmentation. In contrast, genetic variability in populations in southern California is much higher (Gray, 1995), but data are not specifically available for turtles in the WMPA. Given the isolated status of populations in the Mojave River, decreased genetic variability is a possible conservation concern for these turtles.

An Upper Respiratory Disease-like syndrome was responsible for the death of 35-40% of the individuals in one of only two known populations in Washington in 1990. Although not positively confirmed elsewhere, a large die-off of 42 western pond turtles in northern California in 1993 suggests that the disease is not confined to Washington (Holland, 1994).

Other localized threats include contaminant spills, grazing, and off-road vehicle use (Fish and Wildlife Service, 1993), and all are real threats to the continued survival of the species in the Mojave River. Significant portions of the Afton Canyon area occupied by western pond turtles have been negatively affected by grazing and off-road vehicle use.

Invasion of exotic pest species into habitats occupied by western pond turtles is another threat to the continued survival of the species in the WMPA. Saltcedar, or tamarisk (*Tamarix ramosissima*), is an invasive pest plant species that is firmly established in the Mojave River system (Lovich et al., 1994). The changes in channel morphology and hydrology associated with saltcedar invasion in the Mojave River have degraded what little western pond turtle habitat exists in the WMPA (Lovich and de Gouvenain, 1998). Dudley and Collins (1995) suggested that the introduction of non-native turtles including red-eared sliders (*Trachemys scripta*) and painted turtles (*Chrysemys picta*) into California threatens western pond turtles, however no data are available to substantiate this claim. The former species is likely to occur in the WMPA because of its ubiquity worldwide (Ernst et al., 1994), but I am aware of no specific records. Holland (1994) suggested that disease might be spread into western pond turtle populations by introduced turtle species.

The bullfrog (*Rana catesbeiana*) is native to eastern North America, but is widely established in California including the WMPA. As gape-limited predators, bullfrogs will eat any live animal they can swallow and this includes hatchling and young western pond turtles (Holland, 1994). The intensity of predation from bullfrogs is great enough to eliminate recruitment in some western pond turtle populations in southern California (Overtree and Collings, 1997; Robert Goodman, personal communication).

Humans widely utilized western pond turtles for food at least until the 1930's, and exploitation continues on a smaller scale in some areas. Turtles are also collected for resale in the pet trade. Bury (1989) reported that one pet wholesaler obtained about 500 western pond turtles from a southern California lake and shipped them to Europe. Roads also take their toll on western pond turtles in some areas through vehicle strikes (Holland, 1994). Exploitation of western pond turtles in the WMPA for these purposes would be devastating.

Biological Standards:

Protection of the western pond turtle in the WMPA will require habitat protection and restoration in the Mojave River system at the minimum. Effective conservation of aquatic turtles requires recognition of the need to protect adjacent terrestrial habitat for nesting (Rathbun et al., 1992), hibernation, and estivation (Burke and Gibbons, 1995). Potential dispersal corridors among aquatic habitats must also be protected (Holland, 1994). For western pond turtles this may mean protecting riparian corridors 500 m or more from the wetland boundary (Holland, 1994).

Grazing and off-road vehicle use should be eliminated from riparian corridors utilized by this species for feeding, nesting, and overwintering/estivation. New roads should not be constructed through or adjacent to habitats occupied by the turtle to prevent additional mortality from road kills.

Ongoing saltcedar eradication efforts by the Bureau of Land Management at Afton Canyon must be continued along with associated revegetation of the area with native plants to restore degraded habitat. Bullfrog control may be necessary to prevent or eliminate excessive predation on hatchling and juvenile turtles.

Construction of small ponds along the floodplain of the Mojave River seem to attract western pond turtles as shown by the case of Camp Cady. Construction of additional ponds may provide valuable refugia to turtles during times of drought or in areas where the stream channel has been severely degraded by saltcedar.

In light of the observation by Holland (1992) regarding the morphological distinctiveness of western pond turtles in the Mojave River, consideration should be given to the possibility of establishing satellite populations in other protected desert wetlands as a hedge against extirpation from some catastrophe caused by a chemical spill, disease, or other factors. Sites that appear to be suitable include artificial wetlands at the Desert Studies Center at Zzyzx and Dos Palmas Oasis in Riverside County. Similar relocations have been implemented successfully for other aquatic vertebrates in the desert including various pupfish and tui chubs.

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